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The Pest Issue



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Front cover: A mob of Eastern Grey Kangaroos *Macropus giganteus*. Photo Graeme Coulson. See page 251.

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Introduction

Maria Gibson

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Upon occasion, *The Victorian Naturalist* receives articles that can be published together as a special issue relating to a single theme. This serendipity occurred in 2010, when all articles in 127(4) concerned one or more environmental weeds. The present issue is the first that focuses solely on Australia's environmental pest species. *The Pest Issue* was suggested by Dr Desley Whisson from Deakin University, who is thanked for her editorial assistance and efforts in searching out willing authors.

The Pest Issue provides readers new to the topic with an entry into the world of Australian environmental pests and their management, while the expert is alerted to current projects

and practices. Articles are not in any particular order, but are loosely divided into three groups. The sequence begins with four ostensibly miscellaneous papers, each of which demonstrates the human dimensions of pest issues. Whisson *et al.* provide an overview of the history of pest plant and vertebrate introductions and their impacts. For more than a decade, Acclimatisation Societies introduced into Australia, acclimatised and domesticated species deemed useful or desirable; and introduced native species into areas of Australia where they were previously unknown. But, of course, too much of a good thing causes it to lose some, or all, of its gloss. Proliferation of certain plants and

animals caused significant adverse impacts to the natural environment, necessitating eradication and/or management. The paper by Whisson *et al.* is followed by a Naturalist Note. The usual practice for this journal is to begin with Research Reports, followed by Contributions and then to present Naturalist Notes but, for this issue, articles are presented in loose groups with a perceived secondary theme and the reader is trusted to determine the nature of each article. Hubregtse's Note considers *Erigeron karvinskianus*, a common garden plant regarded as an environmental weed in south-eastern Australia. An account of native and introduced arthropods that use this plant as a larder or place of residence is presented, demonstrating that there are, at least, two sides to each story. Presland's article 'On the Peppertree *Schinus L.* in Australia' continues this philosophy. Presland recognises the invasiveness of the species but argues that the historical and cultural importance of particularly significant examples of *Schinus* is embedded in the national psyche, thus, the species merits recognition as more than 'just a weed'. In the final article for this group, Dell reviews a group of plants, bryophytes, usually ignored by the public and environmental managers alike. He considers available literature and provides a list of taxa with weedy characteristics, designating an origin status for each. He provides a case study for examining concepts of bryophyte introductions, using *Pseudoscleropodium purum*, probably the most well-known invasive moss in Australia.

The second and largest group of articles primarily concern management. McArthur's paper sets the scene for this grouping and presents the Victorian Government's current (2020) framework for the management of weeds and pest animals on public land. The following seven articles examine novel methodologies, policy approaches, community approaches, eradication of pests on islands, marine pests, and native animals as pests. Woolnough *et al.* recognised a gap in the Victorian Government's ability to manage non-indigenous birds and describe the construction and implementation of a policy that enables their management. With another group of authors, Woolnough describes a community-led approach to managing rabbits, via the Victorian Rabbit Action Network. Bolden

and Johnston report on the procedures required to eradicate a small Feral Pig population on ecologically-sensitive Quail island, while Johnston *et al.* discuss how the current regulatory environment impacts on a proposal to eradicate Feral Cats on French Island. The paper by Moore takes us to alpine Victoria where willows are problematic. The author describes the research and management efforts taken in order to protect alpine plant communities and habitats from the impacts of these trees. This is followed by a paper concerning a marine pest, the introduced kelp *Undaria pinnatifida*. Pocklington presents a novel, biosecure method of management that involves dissection of the portion of stipe that bears sporophylls. In the final offering in this group, Coulson *et al.* focus on the adverse impact kangaroos have on bandicoots, and remind us that in environmental management, as in life generally, 'Those who cannot remember the past are condemned to repeat it'.

After reading these papers, it will be clear to the reader that effective management of an organism requires an understanding of its ecology. Although the articles of group two are focused on management, most also provide considerable detail of the target organism's ecology, hence their 'loose' grouping. The third group of papers has a more ecological focus, with implications for management of target pests. Hampton and Davis briefly review environmental, agricultural and human health impacts of deer, and recognise deer management as an urgent issue requiring a strategic response. Crockett and Carnell find that native sea urchins have a greater adverse impact on the algal community than the introduced *Undaria pinnatifida*. Pickering and Ansong examine human-mediated seed dispersal of Australian weeds and identify 212 species that have been spread from clothing/personal equipment, as well as 375 spread from vehicles.

The Editors commend this special issue of *The Victorian Naturalist* to our readers. Victoria is a place of unique and diverse environments. We feel that the appreciation and enjoyment of its many natural features can only be enhanced by a greater knowledge of the work that is done to protect and manage those environments.

History of Australian plant and vertebrate pests: introductions and impacts

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Abstract

Australia has a long history of exotic species' introductions, naturalisation and spread, and facilitated range expansions of native species. Together with broadscale landscape modification, this has resulted in an unprecedented number of pest species across a broad range of taxonomic groups (e.g. vertebrate and invertebrate animals, plants, algae and fungi), with impacts on native species, biodiversity, ecosystem function, the economy, and human health. Overgrazing and browsing by introduced herbivores contributes to land degradation. Grazing, predation, and competition by exotic vertebrates threaten many endangered species and communities. Feral predators (i.e. Feral Cat *Felis catus* and European Red Fox *Vulpes vulpes*) are implicated as key contributors to Australia's endemic mammal extinctions. Weeds affect the structure and function of many ecosystems, displace native plant and animal species, harbour pests and diseases, and alter fire regimes. In this paper, we review the history of Australia's pest animals (vertebrates) and weeds, including the reasons for introductions, factors leading to both exotic and native species being pests, and the consequences for Australia's environment and biodiversity. (*The Victorian Naturalist* 137(6), 2020, 162–169)

Keywords: pest species, environmental pest, exotic species, vertebrates, weeds

Introduction

This paper provides an overview of Australia's pest vertebrate and plant history. We highlight why pest impacts on natural ecosystems and biodiversity are important and provide an overview of the long history of introductions of exotic species into Australia as well as the redistribution of native species outside their natural range. The factors leading to a species becoming a pest and the environmental impacts of pests are presented.

A lot to lose

Australia's ancient geology, climate and long isolation from other continents have resulted in a unique biodiversity. Indeed, Australia is considered one of 17 mega-diverse countries (Department of Agriculture, Water and the Environment [DAWE] 2020a). Collectively, mega-diverse countries comprise less than 10% of the Earth's surface but support more than 70% of its biological diversity. There is a high level of endemism within Australia's plants and animals, with many species having unique adaptations to deal with low-nutrient food sources and unpredictable rainfall (Chapman 2009). Approximately 87% of mammals, 45% of birds, 93% of reptiles, 94% of frogs and 92% of vascular plants are endemic (Chapman 2009). High

levels of endemism also are found in freshwater fish (61%) (Hoese *et al.* 2006) and inshore fish species in the southern temperate zone (80%) (Pogonoski *et al.* 2002).

It therefore is concerning that Australia's biodiversity is declining and that the outlook is generally poor due to increasing threats (Creswell and Murphy 2017). Since records have been kept, 54 faunal and 37 floral species have become extinct and a further 464 faunal and 1336 floral species are now nationally listed as critically endangered, endangered, vulnerable or conservation dependent (DAWE 2020b). Invasive species are the second most serious threat to Australia's biodiversity after habitat loss, and their impacts are increasing (The Environment, Communications, Information Technology and the Arts References Committee 2004; Creswell and Murphy 2017).

A long history of introductions

Australia has had a long history of animal and plant introductions. Of around 650 vertebrate animal species introduced to Australia, 81 have established in the wild, with around 30 of these considered pests (Long 1981; Bomford 2008; Invasive Plants and Animals Committee [IPAC] 2017a). Over 27 000 vascular plant

species have been imported into Australia (Virtue *et al.* 2004) with approximately 7% (1765 species) considered pests in natural ecosystems (Groves *et al.* 2005).

The first terrestrial animal to be introduced to Australia was the Dingo *Canis lupus dingo* but its 'pest' status is hotly debated (Allen *et al.* 2013). The Dingo arrived from South-East Asia with the Toolean people from south Sulawesi between 5000 and 10000 years ago (Fillios and Taçon 2016). As a large carnivore, the Dingo may have had considerable impacts on Australia's native fauna. Indeed, this predator has been implicated as the primary cause for the extinction of the Thylacine *Thylacinus cynocephalus* and the Tasmanian Native-hen *Tribonyx mortierii* from mainland Australia (Baird 1991; Fillios *et al.* 2012). However, the Dingo is now considered to be a functional part of the ecosystem as a top-order predator, suppressing populations of herbivores (e.g. kangaroos) and introduced mesopredators (Feral Cat *Felis catus* and European Red Fox *Vulpes vulpes*) (Glen *et al.* 2007).

Another early animal introduction was the Pacific Rat *Rattus exulans*, a species that is native to south-east Asia. It was introduced to Norfolk Island around 900 years ago, possibly by Polynesians who valued it as a food source (Watts and Aslin 1981; Anderson and White 2001). Pacific Rats have since been recorded from Adele Island in northern Western Australia (Taylor and Horner 1973), Christmas Island (Watts and Aslin 1981) and the Murray Islands (east of Torres Strait) (Watts and Kemper 1989). On Norfolk Island, the Pacific Rat has been implicated in the local extinction of the Providence Petrel *Pterodroma solandri* and Pycroft's Petrel *Pterodroma pycrofti*, and the decline of the Lord Howe Island Gecko *Christinus guentheri* (Cogger *et al.* 1993).

The first known plant to be introduced and become naturalised in Australia was Tamarind *Tamarindus indica*. This species was introduced by the Macassans on their voyages to northern Australia to collect trepang (also known as sea cucumbers) in the 1700s (Groves *et al.* 2005). Tamarind is now naturalised, spreading, and regarded as an environmental weed in northern Queensland (Queensland Government 2016).

The arrival of Europeans in Australia in the 1700s marked the beginning of an era of numerous introductions of exotic plant and animal species. Explorers planted fruits and vegetables and released chickens and Pigs *Sus scrofa* in the hope that they would establish populations that would become a source of food for future expeditions. The early explorers also may have been responsible for the unintentional introduction of Black Rat *Rattus rattus* and House Mouse *Mus musculus* that were stowaways on their ships. Then came the First Fleet and the British settlers who introduced numerous plants for crops, pasture, forestry, gardens, and by accident (Virtue *et al.* 2004; Groves *et al.* 2005). Vertebrate animals were introduced for transport and work, food and animal products, hunting and fishing, for biological control, as pets, for their aesthetic value, and by accident. Many vertebrates either escaped captivity or were deliberately released.

In the 1800s, the establishment and spread of many plant and animal species was encouraged. In Victoria in 1857 and 1858, Ferdinand von Mueller, a government botanist and the director of the Royal Botanic Gardens, Melbourne, was solely responsible for distributing 7120 living plants and 22438 packets of seed throughout the Victorian colony (Tout-Smith 2003). The European Rabbit *Oryctolagus cuniculus*, one of Australia's worst vertebrate pests, was introduced in 1859 by settler Thomas Austin to his property near Geelong (Rolls 1969).

Acclimatisation Societies, established in 1861 (Victoria and New South Wales) and 1862 (Queensland, South Australia, and Tasmania), played an active role in importing, breeding, and releasing exotic plants and animals in Australia for about a decade. Their mission was to introduce, acclimatise and domesticate useful or ornamental exotic species; and to spread indigenous animals from parts of the colonies where they were already known to other localities where they were not known (Gillbank 1984). Collectively, these societies attempted to introduce at least 95 vertebrates (Long 1981; Bomford 2008; Braysher 2017). The Acclimatisation Society of Victoria was responsible for the successful introduction of three major vertebrate pests (European Starling *Sturnus vulgaris*, House Sparrow *Passer domesticus*,

and European Carp *Cyprinus carpio*), and numerous plants that are now considered weeds. However, approximately 60% of vertebrate introductions by the Acclimatisation Societies failed (Rolls 1969). For example, at least 1000 mongooses *Herpestes* spp. were released at 14 locations in south-eastern Australia between 1855 and 1883, primarily to help control European Rabbits (Peacock and Abbott 2010). Failure of mongooses to establish may have been due to their destruction by 'rabbiter' who made a living from trapping European Rabbits, or it may have been a poor climate match.

Private citizens, industry, zoos, and governments continued to release exotic animal and plant species once the Acclimatisation Societies folded. Some species were accidental escapees from captivity (e.g. Grey Squirrel *Sciurus carolinensis*, Peacock 2009) but others were deliberate releases for biological control (e.g. Cane Toad *Rhinella marina*), or species that were released once they were no longer needed or valued (e.g. Dromedary *Camelus dromedarius*, Horse *Equus caballus*, and deer *Cervus* spp.). Most plants were introduced for ornamental horticulture; approximately 94% of all exotic plants introduced are found in home gardens (Virtue *et al.* 2004; Groves *et al.* 2005). Of an estimated 27 009 plant species introduced to Australia, only 4% appear to have been accidental (e.g. contaminants of grain), with 7% introduced for food crops, pasture, or forestry (Groves *et al.* 2005). The rate of plant species introductions appears to have been relatively steady since 1880 (Dodd *et al.* 2015).

In the 1800s, the establishment and spread of some vertebrate pest species was facilitated by vegetation clearance and modification, pasture and crop establishment, and the provision of watering points. European Rabbits benefitted from pasture establishment and the availability of burrows formerly occupied by wombats (Myers 1986). European Red Foxes thrived due to good hunting opportunities on agricultural land, and increased food availability from the spread of European Rabbits (Abbott 2011). Improved pasture and provision of water points for stock probably aided the successful establishment of species including the Donkey *Equus asinus* and Goat *Capra hircus*. The persecution of native species (e.g. Wedge-tailed Eagle

Aquila audax, quolls *Dasyurus* spp, Thylacine) that were perceived as threats to livestock, also may have contributed to the success of some vertebrates including European Rabbits and Feral Cats (Abbott 2002; Abbott 2008; Peacock and Abbot 2013).

From introduction to pest

The Australian Pest Animal Strategy (IPAC 2017a, p. 4) defines a pest animal as a species 'that causes more damage than benefits to human valued resources and social wellbeing'. It addresses only vertebrate pest species. In the Australian Weeds Strategy (IPAC 2017b, p. 4), a weed is defined as 'a plant that requires some form of action to reduce its harmful effects on the economy, the environment, human health, and amenity'. Less than 10% of plant and animal species that have been introduced to Australia have become pests that impact the natural environment (Bomford and Hart 2002; Virtue *et al.* 2004). Figure 1 shows the percentage of established environmental weeds in Australia (N = 1765) that were introduced for food, pasture, forestry, gardens and by accident; Figure 2 shows the percentage of established vertebrate pests in Australia (N = 81) that were introduced for transport or work, food and animal products, hunting or fishing, biological control, as pets, for their aesthetic value, and by accident.

Some plant species were identified as pests soon after their introduction, and efforts were made to eradicate them. For example, Victoria's *Thistle Prevention Act* was passed in 1856, to provide for the eradication of four particularly problematic thistles and the Bathurst Burr *Xanthium spinosum* (*Thistle Prevention Act 1856*).

Analysis of the characteristics of vertebrates that have become pests suggests that a good climate match between their native range and Australia was a key factor in their establishment and spread (Forsyth *et al.* 2004). In addition, vertebrate pests in Australia typically have a history of establishment in other countries where they have been introduced, a high reproductive rate, a generalist diet, and an ability to live in human-disturbed habitats (Forsyth *et al.* 2004).

A multitude of extrinsic and intrinsic characteristics have been identified as facilitating plant species becoming pests in Australia (see

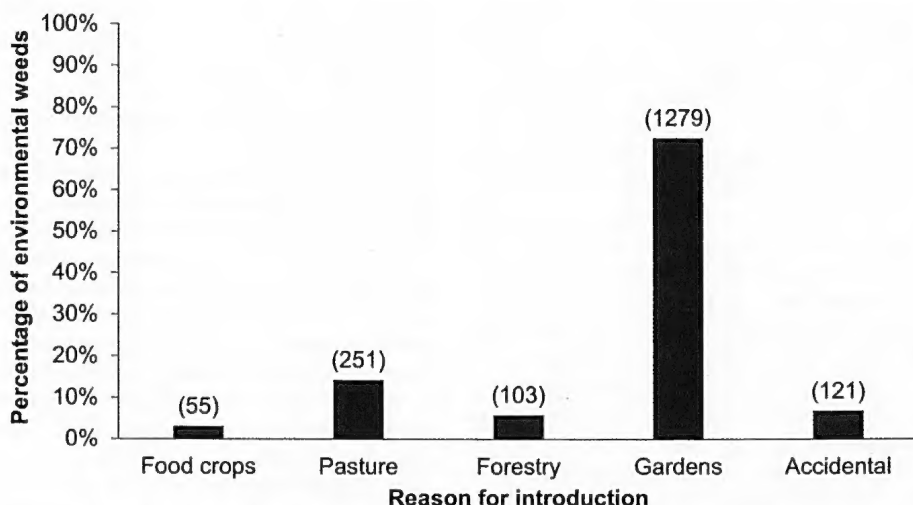


Fig. 1. Percentage of established environmental weeds in Australia (N=1765) introduced for food, pasture, forestry, gardens and by accident (values obtained from Virtue *et al.* 2004). Some species were introduced for multiple reasons, so the sum adds to greater than 100%. The number of species is indicated above the bars.

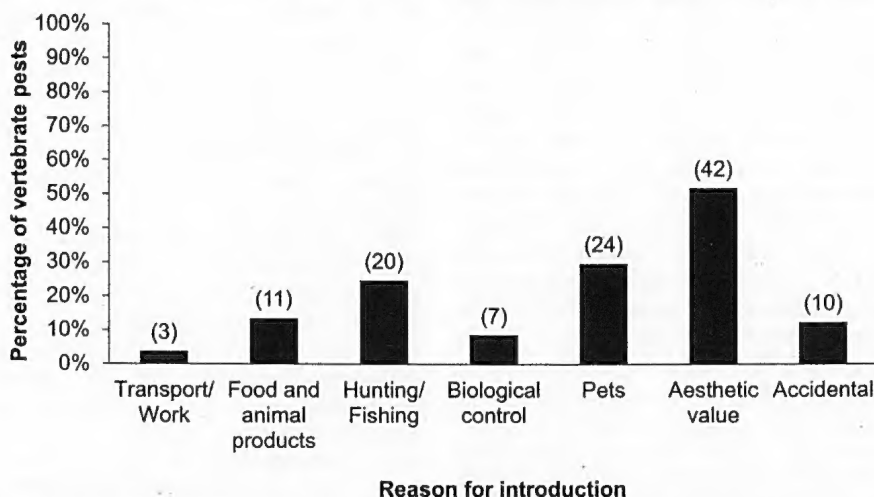


Fig. 2. Percentage of established vertebrate pests in Australia (N = 81) introduced for transport or work, food and animal products, hunting or fishing, biological control, as pets, for their aesthetic value, and by accident (compiled from lists in Long 1981, Bomford 2008 and Braysher 2017). Some species were introduced for multiple reasons, so the sum adds to greater than 100%. The number of species is indicated above the bars.

Gibson 2010 for review). Introduction effort, climate match, and introduction into disturbed environments are key extrinsic factors (Virtue *et al.* 2004). Intrinsic characteristics include being tolerant or adaptable to a wide range of climatic and soil conditions; having rapid growth rates; high reproductive capacity; effective dispersal

mechanisms; and possessing other characteristics (e.g. allelopathy) that give them a competitive edge over native species (Gibson 2010).

Native species as environmental pests

Some native species also have become pests due to range expansions (usually facilitated by

people), or landscape changes that facilitate an increase in abundance of the species. In the 1800s, native animals were translocated for a variety of reasons: in Western Australia, the Laughing Kookaburra *Dacelo novaeguineae* was introduced for snake control, and the Sulphur-crested Cockatoo *Cacatua galerita* was introduced for keeping as pets. Both species have spread and now compete with local species for nest hollows. Since the late 1800s, many translocations have been undertaken for conservation reasons, with islands considered as conservation arks (Burbidge *et al.* 2018). For example, in 1923 Koalas *Phascolarctos cinereus* were translocated outside of their native range to Kangaroo Island, South Australia, to protect them from hunting that threatened their populations on the mainland (Whisson and Ashman 2020). These Koalas thrived and became a major pest problem on the island (Masters *et al.* 2004).

Native plants have been translocated, usually for use in gardens. Sweet Pittosporum *Pittosporum undulatum*, originally confined to rain-forest gullies in south-eastern Victoria, New South Wales, and Queensland, was introduced to gardens throughout Australia. It has invaded and transformed many natural ecosystems and is now considered a major woody weed outside of its natural range (Blood 2001). The Cootamundra Wattle *Acacia baileyana*, originally from central western NSW, and extensively planted in gardens and shelterbelts, also has become a major environmental weed in south-eastern Australia. It is a serious threat to many heathland, woodland and forest communities, where it outcompetes and impedes germination of local native species (Carr *et al.* 1992). Cootamundra Wattle also hybridises with at least six other Australian wattles, thereby threatening the genetic integrity of native wattle populations (Robinson 2007).

Some native species have become pests due to landscape change and an increase in favoured habitats or food resources. In the arid interior, the provision of permanent watering points for stock has resulted in an increase in abundance of some kangaroo species, as well as parrots. Woodland and forest clearance for agriculture and urbanisation has resulted in forest fragmentation and an increase in the edge habitat preferred by Noisy Miners *Manorina*

melanocephala. This species continues to colonise more and more habitat, to the exclusion of many other native species (Clarke *et al.* 2007).

Environmental impacts of weeds and vertebrate pests

Pest plants and vertebrates cause a wide variety of impacts on the natural environment, economy, and people (Hart and Bomford 2006; Hone 2007). Although there have been some attempts to quantify the impacts of pests in Australia, they focus on costs to agriculture or livestock production and provide only conservative estimates of environmental impacts based on the cost of management (Bomford and Hart 2002; Sinden *et al.* 2004; Gong *et al.* 2009). For example, Bomford and Hart (2002) reported annual management costs of \$20m for European Rabbits, \$7m for European Red Foxes, \$2m for Feral Goats and \$1m for Feral Cats. Reddiex *et al.* (2006) estimated that \$21.3m was spent on labour costs alone for European Red Fox control in Australia from 1998 to 2003. Sinden *et al.* (2004) reported that \$19.6m was spent on weed control in National Parks and National Heritage Trust areas in 2001–2002. Using Australian Bureau of Statistics consumer price indices between 2002 and 2018, this cost inflated to \$29m for 2018 (McLeod [eSYS Development Pty Limited] 2018).

Vertebrate pests threaten biodiversity through their predation on and competition with native species, and by transmitting disease. They also degrade habitats through their feeding or trampling, overgrazing and overbrowsing, and by spreading weeds. Further, there is an increasing understanding that co-occurring pest species can exacerbate impacts. For example, European Rabbits provide a food source to European Red Foxes and Feral Cats and thus contribute to an increase in predator populations and hyperpredation of native species (Courchamp *et al.* 2000). The impacts of pest animals also may be influenced by other factors such as climate or fire. For example, pest herbivores such as deer and Horses may threaten regeneration of native plants post fire (Giljohann *et al.* 2017).

Almost all of Australia's native vegetation communities (including those within sites of global or national significance) have been invaded or are vulnerable to invasion by weeds

(Csurhes and Edwards 1998). Weeds compete with native plants for space, nutrients, moisture, and sunlight, altering hydrological cycles and fire regimes, and transforming ecosystems through changes to vegetation structure and plant species composition (Sinden *et al.* 2004). Weed invasion can have significant impacts on fauna that rely on native vegetation communities for food and shelter. Species that cannot adapt to the modified habitat must disperse if they are to survive. However, weed invasion also can have positive benefits for some species. For example, Blackberry *Rubus fruticosus* provides habitat for some native small mammal and bird species and therefore can increase diversity in an area (Packer *et al.* 2016). African Boxthorn *Lycium ferocissimum* provides food resources for Singing Honeyeaters *Lichenostomus virescens* in a coastal wetland (Carlos *et al.* 2017). Removal of Gorse *Ulex europaeus* and Hawthorn *Crataegus monogyna* was shown to decrease bird diversity in a revegetation program (Carlos and Gibson 2010).

Managing pests

In recent decades there has been a shift in pest management from reactive and relatively *ad hoc* single-species management, to a more holistic ecosystem-level approach that recognises pest species as part of a complex web of species interactions, and the roles of the stakeholders involved (Johnson and Charlton 2010; Braysher 2017). Research into the environmental impacts of pests and action to address those impacts also has increased (Whisson 2010).

The development of National Pest Strategies for Animals and Weeds in 2007 (with revision in 2017) were major improvements in pest management in Australia, emphasising the need for a biosecurity approach, providing guidelines for best practice management of established pests, and recognising the challenges to effective pest management (IPAC 2017a, 2017b). As part of this approach, the Weeds of National Significance program identifies 32 weeds based on their invasiveness, potential for spread, and their environmental, social and economic impacts. In addition, another 28 non-native weeds that are in the early stages of establishment are identified on the Alert List for Environmental Weeds. These weeds could become a significant

threat to biodiversity if they are not managed. States also have their own legislation and policy frameworks. In Victoria, introduced plant and animal pests are classified under the *Catchment and Land Protection Act 1994*. Under this Act, landowners are legally obliged to manage noxious weeds and pest animals on their land.

Concluding comments

Australia's long history of pest introductions has had irreversible consequences for our natural environment. Decisions and actions of the past have resulted in pest populations that are now widespread and embedded in ecosystems. These pests will require ongoing management to minimise their impacts. We have learnt the hard lesson that introducing exotic species should not be taken lightly, and that we must be vigilant to prevent new pest incursions.

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Erigeron karvinskianus: friend and foe

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Erigeron karvinskianus (Fig. 1), also known by names such as Mexican Daisy, Seaside Daisy and Karwinsky's Fleabane, is a vigorous plant belonging to the family Asteraceae. It was first described in 1836 by Swiss botanist Augustin Pyramus de Candolle. The genus name comes from two Greek words, 'eri' meaning early + 'geron' meaning old man, referring to early flowering plus the white down that is characteristic of some *Erigeron* species (Collins Dictionary 2020). The specific epithet refers to Wilhelm Friedrich Karwinski von Karwin, who collected the plant in Mexico (Centre for Agriculture and Bioscience International [CABI] 2019).

Erigeron karvinskianus is native to Mexico, Honduras, El Salvador, Guatemala and Colombia, but occurs as a weed in many subtropical

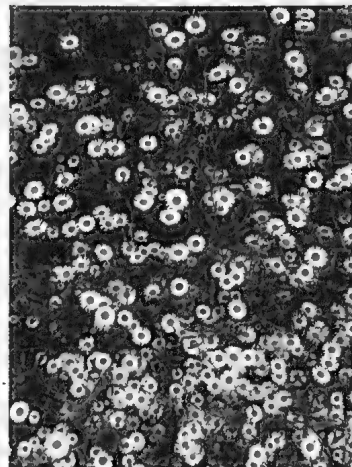


Fig.1. *Erigeron karvinskianus*.

and temperate regions of the world. It is now found in North and South America, the West Indies, southern and western Europe, Africa, Indian Ocean islands, Asia, Australia, New Zealand and Pacific islands. It is listed as a major invasive plant in tropical islands in the Indian and Pacific Oceans, Africa, Australia and New Zealand, and was first recorded in Australia in 1908 (CABI 2019).

This plant has dainty daisy flowers with a radiate head of outer ligulate florets and inner tubular florets. It blooms for most of the year, attracting butterflies and other insects. It can grow in almost any habitat, reproduces rapidly, has no known natural enemies, and is disease free (CABI 2019). No wonder it appeals to gardeners around the world! In Britain, it has been given the Royal Horticultural Society's Award of Garden Merit (Wikipedia 2020) and is promoted as a hardy garden plant.

I am guilty of growing *E. karvinskianus* in my garden in the Melbourne suburb of Notting Hill. Several years ago, I collected a number of small plants from a nearby roadside and planted them next to the front footpath, in a bare strip of ground where nothing else would grow. They thrived, even in hot, dry weather, and it was not until the 2019/2020 summer that some of them died. By this time, they had become well established in the back garden, where they flourish in a partly shaded spot near the south side of the house.

Although its vigorous growth and tendency to smother other plants make *E. karvinskianus* an undesirable addition to any Australian garden, this invasive weed is used by many arthropods, both native and introduced, as a place of residence and as a larder. Between 2016 and 2019, it attracted many such creatures, some of which I hadn't seen elsewhere. Spiders, a katydid, plus a variety of flies, bugs, beetles, bees, wasps, butterflies and moths all made use of this plant.

The most common spiders were young lynx spiders (Oxyopidae), which waited on the flowers for prey to arrive (Fig. 2), and hid behind their chosen flower when disturbed. There were also tiny flower spiders (Thomisidae), and a larger spider that I watched while it consumed its web (Fig. 3). A green katydid nymph (Tettigoniidae) with its characteristic long antennae (Fig. 4) was seen only once. Introduced small



Fig. 2. Lynx spider waiting for prey.



Fig. 3. Spider consuming its web.



Fig. 4. Katydid nymph.

greyish bugs in the family Lygaeidae were prevalent in late summer and early autumn. Nothing seems to eat these bugs, so they proliferate unhindered. There were only a few beetles, including Australian Carpet Beetles *Anthrenocerus australis* with blotchy brown and buff upper surfaces (Fig. 5). Flies of many shapes, sizes and colours were abundant. They included Common Drone Fly *Eristalis tenax* (Fig. 6), bee fly (Bombyliidae) (Fig. 7), three different sorts of hover flies (Syrphidae) (e.g. Fig. 8), and many other flies, some of which were very tiny—about half the size of a single ligu-



Fig. 7. Bee Fly.



Fig. 5. Australian Carpet Beetles.



Fig. 8. Hover Fly.



Fig. 6. Common Drone Fly.



Fig. 9. Looper caterpillar.



Fig. 10. Greenish Grass-dart.



Fig. 11. Wasp.



Fig. 12. Native bee.

late floret. I saw no moths, but their offspring in the form of looper caterpillars were present, some green (Fig. 9) and some yellowish. Butterflies seen included Common Grass Blue *Zizina labradus*, Greenish Grass-dart *Ocybadistes walkeri* (Fig. 10) and the introduced Cabbage White *Pieris rapae*. Wasps (Hymenoptera) (e.g. Fig. 11) were less common, but some small bees (Hymenoptera) (e.g. Fig. 12) came to the flowers quite frequently. Honeybees *Apis mellifera* visited the flowers occasionally, but didn't seem able to gather much from them.

The largest variety of arthropods was present in 2016. Since then, and particularly since 2018, the number and variety of creatures visiting or living on the plants has diminished alarmingly. For more than two and a half years from early 2018 I found no spiders, although two small ones were present in late September 2020. Even the Lygaeidae bugs are less abundant. Since *E. karvinskianus* plants have provided food and shelter for so many creatures in the past, it seems clear that biodiversity (particularly among the invertebrates) is declining in this area, as it is worldwide.

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On the peppertree *Schinus* L. in Australia

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Abstract

Two species of *Schinus*, a genus in the family Anacardiaceae, were introduced to Australia in the 19th century. Both *S. molle* and *S. terebinthifolia* are native to parts of South America but have naturalised and become invasive weeds in this country. Although both species have caused environmental damage in a number of areas, they are regarded with some affection by many people. *Schinus molle*, in particular, has become an entrenched botanical feature in numerous rural and urban settings and has achieved recognition as a significant part of the social life and the history of many local Australian communities. (*The Victorian Naturalist* 137(6), 2020, 173–178)

Keywords: Peppercorn Tree, Anacardiaceae, *molle*, botanical history

Introduction

The peppertree *Schinus* L. has an interesting place in the botanical history of Australia. Imported more than 160 years ago for its shading qualities, and widely planted as an ornamental, today it is regarded in most states as an invasive weed. The tree is not unique in this regard, of course; there are many such species. Indeed, it has been estimated that at least one-third of the species that have been declared noxious in Australia were introduced for their ornamental value (Panetta 1995). However, *Schinus molle* is the only exotic weed species which includes numerous examples that are recognised as being significant for historical, social or aesthetic reasons. On these grounds, it can be reasonably argued that to consider this tree species as nothing more than an invasive weed is to ignore its well-documented presence within the fabric of Australian life and history.

The species

The genus *Schinus* L. is in the family Anacardiaceae, and comprises 28 species, all native to the South American continent. There are about 80 genera within the Anacardiaceae, including several economically important species, notably cashews (in the type genus *Anacardium*), mangoes, and pistachios.

Of the 28 species of *Schinus* only two—*S. molle* (Fig. 1) and *S. terebinthifolia*—have been cultivated and become naturalised outside their native regions (Jessup 2020). Although these two species are native to different areas of South America—*S. molle* comes

from Peru; *S. terebinthifolia* from Brazil—both were introduced into Australia, probably separately, in the 19th century, and have since become a familiar sight in many parts of the country.

The generic name comes from the Greek word, 'schinos', which refers to *Pistacia lentiscus*, through perceived similarities in the two genera. The species name *molle* is derived from 'mulli', the term for the tree in Quechua, an indigenous language of Peru. The name *terebinthifolia* refers to a visual similarity of leaf structure in that species with that of the Turpentine Tree *Pistacia terebinthus*. In Australia, *S. molle* is commonly called Peppercorn Tree, Peppertree or Peruvian Peppertree, and sometimes Californian Peppertree; *S. terebinthifolia* is known as the Broad-leaved Peppertree, or Brazilian Peppertree. The scientific name of the latter species has been very commonly misspelled as '*S. terebinthifolius*', through confusion as to the correct gender of the genus. In 2015 it was determined that *Schinus* is feminine, and adjectival names should be spelled accordingly. (Wikipedia 2020a; Zona 2015). A further confusion lies with the practice of referring to the Peruvian Peppertree in Australia as '*S. areira*'. In the 1990s it was decided that the epithet '*areira*' is more accurately applied as a descriptor of variety, e.g. *S. molle* var. *areira*.

Introduction to Australia

A species that is so deeply embedded in the national psyche as the peppertree might easily be assumed to be native to Australia, particularly since records of its introduction are vague or lacking. So, how long have these two *Schinus* species been in Australia? And how did they arrive here? Regarding the first question, there are suggestions that *S. molle* had been planted in both New South Wales (Hambrett 2018) and Western Australia (Steinbauer and Wanjura 2002) by the 1830s. However, it is difficult to assess the veracity of these claims, because in neither case was a source for the information provided.

Certainly, questions as to when and how *Schinus* came to Australia have no simple or unequivocal answer. Perhaps the best suggestions were made by noted naturalist Crosbie Morrison. Considering the presence of mature specimens of *S. molle* in the outback, and its spread to many parts of the continent, Morrison suggested that it was likely peppertree was introduced in the 1850s (Morrison 1956). He further reasoned that the tree was imported from the United States by immigrants during the gold rush period. This theory fits with the fact that, although not native to the area, the tree was known in California during the late 1840s gold rush in that part of the United States.

Following the Spanish invasion and eventual conquest of Central and South America in the 16th century, *S. molle* was taken to Mexico, where it became naturalised. From Mexico it was taken to California (also part of the Spanish empire at the time), probably late in the 18th century. It was planted widely as an ornamental, and spread over a large part of south-western USA (Kramer 1957). By the 1850s *S. molle* was naturalised, and widely available for sale from plant nurseries; it was subsequently used in street plantings in Californian cities (Howard and Minnich 1989).

In the case of *S. terebinthifolia*—also naturalised in California by the 1850s—plants were being offered for sale in Sydney in April 1851 (*Sydney Morning Herald* 1851). This was at about the time that gold was first discovered in significant quantities at Orange in New South Wales, and a few months before large numbers of gold-seekers began flooding into the

Ballarat and Sandhurst districts in Victoria. It is likely, then, that the introduction to this country of *S. terebinthifolia* may have predated that of *S. molle*.

The state nurseries and Botanic Gardens played a major role in the spread of *Schinus* in Australia. In the 1860s, towns in the far west of New South Wales were being supplied with peppertree that had been propagated in the Adelaide Botanic Gardens. When introduced to the Adelaide area, the trees had thrived in the alkaline soil and George Francis, the Gardens' director, set about promoting the species to other dry parts of the nation (Armstrong 1989; Hambrett 2018). By the late 1860s there were mature *Schinus* (probably *S. molle*) growing in the Melbourne Botanic Gardens (*The Argus* 1869). In the 1870s, Francis's counterpart at the Botanic Gardens in Melbourne, William Guilfoyle, regularly included *S. molle* in his landscape designs for public parks. For example, his 1878 design of the park surrounding Lake Weeroona in Bendigo included five peppertrees (Andrews 2012). By 1889, *S. molle* had been planted as an ornamental at the Alice Springs Telegraph Station (Mitchell 1978).

In the first decades of the 20th century, state nurseries in New South Wales regularly distributed peppertrees by the thousand. In 1902, about 2600 specimens were distributed from Gosford alone (Frawley 2009). Many of these trees were bound for the playgrounds of schools, in all parts of rural New South Wales (Hamblett 2018).

In urban environments, *S. molle* was planted for shade in many school yards, but also as a preferred ornamental in parks and as a street tree. It was planted commonly also in private gardens, where it was valued not only for its ornamental qualities, but also because it survived where many other exotic species did not (Head *et al.* 2004). There is evidence from an archaeological investigation at Casselden Place that specimens of *S. molle* were growing in the eastern side of Melbourne's CBD in the latter half of the 19th century (Simons and Maitri 2006).

Once established, *Schinus* reproduces mainly by seeds, which are dispersed

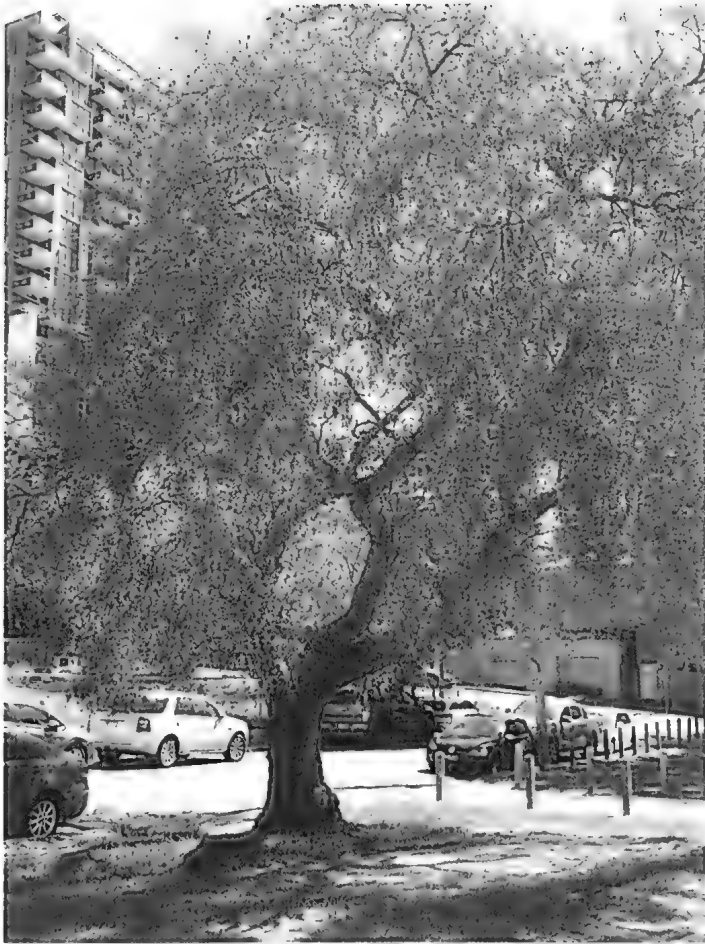


Fig. 1. Peppercorn Tree *Schinus molle*, Box Hill Gardens, Victoria.

primarily by birds, and also other animals, that eat the tree's fruit (Panetta and McKee 1997). In the Adelaide area, Mistletoebirds *Dicaeum hirundinaceum*, for example, have been recorded feeding on the berries of peppertrees (Reid 1983). Coulson *et al.* (2014) noted that roadside populations of *S. molle* are strongly associated with the presence of large, old trees nearby, mainly *Eucalyptus* spp., which provide perching sites for such frugivorous species. *Schinus* also can spread vegetatively by suckering (Business Queensland 2020).

Uses

In their countries of origin, *Schinus* spp. continue to provide a number of useful products.

In Brazil, the leaves and bark of *S. terebinthifolia* have been put to a variety of medicinal uses, ranging from antiseptic poultices to cures for 'inertia of the reproductive organs, skin complaints, chills, tumors, lymphatic swellings, diarrhea ...' (Morton 1978, p. 355). In Peru, the resin that exudes from the bark of *S. molle* was used as an astringent to strengthen the gums. The fruit of *S. molle* has been documented as a source material for alcoholic fermentation, in ancient Peruvian sites (Goldstein and Coleman 2004) and, more recently, as a condiment (Zahed *et al.* 2011).

Although these qualities of *Schinus* are often cited in published accounts, particularly newspapers (e.g. Anon 1872; Anon 1879; Anon 1883), there are few records of any such use being initiated in Australia. This may be a result of the multi-stage movement of the species away from its native region. *Schinus* was transported from Chile to Mexico, to California, and then to Australia. Kramer (1957, p. 325) suggested that, as a result of these movements, the use of peppertree 'becomes markedly less intensive in each one of these stages of dispersal'. He notes that this is characteristic also of other plants that have been transported away from the region in which they are indigenous.

One of the more noticeable features of *S. molle* is the gum which is exuded through the bark. This is a common feature of many species in the family Anacardiaceae; perhaps the best example is the resin obtained from the Mastic Tree *Pistacia lentiscus*, which was the major component of chewing gum. (Wikipedia 2020b). The resin from *S. molle* has many of the qualities necessary for the manufacture of mastic, but not sufficient to make it economically viable (Anon 1931).

Inter-species associations

Notwithstanding its status as an exotic, *S. molle* is reported to have associations with a number of indigenous insects and a few mistletoe species.

Insects

Agrianome spinicollis, a species of longicorn beetle, has been recorded using *S. molle* as a larval host (Hawkeswood *et al.* 1997). The larval stage of the species is a grub called Poinciana Borer, which on hatching burrows beneath the bark, eating the cambial layer as it goes (Brisbane Trees and Gardens 2020).

The peppertree is noted also as playing host to the larvae of the Emperor Gum Moth *Opodiphthera eucalypti*. Well-known naturalist Donald Macdonald, writing in *The Argus*, suggested that this association was formed 'as soon as the Pepper tree (*Schinus molle*) was introduced' (Macdonald 1917, p. 5). Macdonald was writing about specimens in southern Queensland, but almost 25 years before, Charles

French, Victorian Government Entomologist, and founding member of the Field Naturalists Club of Victoria (FNCV), had noted the same inter-species association 'in many parts of the colony' (Anon 1893). Macdonald also believed that the caterpillars were never so numerous as to be harmful to the trees. Alfred Hardy, another FNCV stalwart, remarked that the larvae 'feeds indifferently on the native eucalypts or the introduced Peruvian "Peppertree"' (Hardy 1915, p. 111).

Mistletoes

In an inventory of aerial mistletoe species (Downey 1998), *S. molle* was listed as commonly being used as a host by four species of mistletoe. These plants included *Dendrophthoe glabrescens*, *D. vitellina*, *Muellerina eucalyptoides* and *M. celastroides*. The last species, commonly called Banksia or Coast Mistletoe, was originally described and named as *Loranthus celastroides* (Wikipedia 2020c). A specimen growing on a limb of a *S. molle* was exhibited using that name at an FNCV meeting in February 1915 (Anon. 1915). It was remarked there that since no other mistletoe was growing within about three miles (4.8 km) of the site, the seed must have been carried some distance by a bird.

Weeds versus history

Schinus terebinthifolia is currently naturalised in Western Australia, Queensland and New South Wales (Weeds Australia 2020). It is considered one of the most invasive naturalised plants in south-eastern Queensland, where it is a restricted plant under the *Biosecurity Act 2014*, and an increasing problem in north-eastern NSW (Hosking *et al.* 2003).

In Victoria, *S. terebinthifolia* is not known to be naturalised, but both *Schinus* species are listed as invasive (Victorian Resources Online [VRO] 2020). In parts of New South Wales and southern Queensland, the Broad-leaved Peppertree forms dense thickets, which shade out and smother native vegetation, and destroys the habitat for animals that depend on native plants (Business Queensland 2020). The presence of Peruvian Peppertree can potentially 'alter the structure and composition in grasslands, woodlands and coastal scrub areas'

(VRO 2020). In urban settings, *S. molle* can cause problems to nearby structures because it has a very vigorous root system. Peppertree roots have been recorded stretching more than 50 m in order to find water and nourishment. The vigorous roots need to be kept away from pipes, drains, paths and foundations because of the damage they can cause (alpine treeremoval 2020).

Despite the invasive nature of both species of peppertree in this country, and the detrimental impact this brings to local environments, for many people there is an enduring attachment, affection even, for the *Schinus*. It is clear there is more to be said about peppertrees than merely commenting on their nuisance value (Blood and Cuffley 2006). One measure of the central place that the trees have come to occupy in the social and historical fabric of Australian culture is the inclusion of many specimens in state and national heritage registers. The Australian Heritage Database, maintained by the Federal Government, includes 26 entries for heritage-related places, in all of which an individual peppertree or a grove of them is a contributing component of the site's significance. In a few cases the tree is the defining feature. One such entry is for the Peruvian Peppertree planted in Albany, Western Australia in 1886 to mark the birth of a son to a prominent family. It stands today as the last of the street trees which originally lined Grey Street (Department of Agriculture, Water and the Environment 2020).

Similarly, the Victorian Heritage Database (Heritage Council Victoria) lists 70 places in which *S. molle* features in the statement of significance. This number includes the 19 specimens registered in Victoria by the National Trust. The Trust's database also lists six significant *Schinus* in South Australia, and one in the ACT (National Trust 2019).

These trees may be significant in their own right, as in the case of the Albany tree mentioned above. But, more often, they are regarded as mnemonic markers of a wide range of events or places. One such marker is the *S. molle* planted in Mattingley in 1885, to commemorate the opening of the nearby Bacchus Marsh railway station. Where a particular location has been recognised as significant and worthy of heritage protection, frequently there

are individual features of the place, including trees, which contribute to the significance of the site (Dyson 2012). Examples abound in Melbourne of peppertrees that stand as social markers: the *S. molle* within the garden at Bishopscourt in East Melbourne, for instance, are considered significant as characteristic of 19th century garden plantings (Dwyer 2002); and the peppertrees planted in proximity to the Newmarket saleyards serve as reminders that the area was once the scene of essentially rural activities.

Such trees transcend the boundary between natural history and human history. In so doing they assume a range of new meanings and values, which, in turn, demands changes in the way the trees are interpreted. To view *Schinus* as simply a pest plant or an invasive species, is to see only part of the story. By ignoring any historical, aesthetic, or social value the trees might have accrued by virtue of their contexts, is to diminish the close links that exist between nature and culture.

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Bryophytes as weeds in Australia

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Abstract

Some bryophyte taxa in Australia exhibit weedy characteristics and have been considered variously in literature for their adventive distributions following European occupation. A total of 29 taxa are considered here. These include 13 taxa that are regarded as exotic throughout Australia. Two taxa have a reported cosmopolitan distribution, while 14 have cryptogenic or dual origin status but are not suspected to be native to all or part of their current range within Australia. The basis for treating some taxa as introduced is subjectively based on their pattern of distribution outside Australia and their association with anthropogenic habitats within Australia. Factors for consideration regarding the detection and reporting of adventive bryophytes are discussed. Their impact on Australian ecosystems is largely speculative. In the absence of quantitative research there is, unsurprisingly, little attention given to the need or methods for their management. (*The Victorian Naturalist* 137 (6), 2020, 179–194)

Keywords: bryophytes, introduced species, weeds, *Pseudoscleropodium purum*

Introduction

Almost all bryophyte species in Australia are regarded as native, i.e. they occurred in Australia prior to European occupation (1788 for this review). There are many more exotic taxa reported in Australian vascular plants, with at least 3181 species (12.2% of the flora) comprising naturalised introductions (Fensham and Laffineur 2019). It is difficult to determine origin and define weeds when considering bryophytes. For vascular plants, definitions vary to the extent that it is necessary to apply context; for instance, agricultural, ecological or amenity impact. A simple definition of a weed is a plant growing where it is not wanted and, for ecological considerations, this mainly concerns their impacts on native species or ecosystems. Unfortunately, for bryophytes it most often is not clear what these impacts are, so their weed status seldom is identified. The aim of this review is to 1) provide a list of bryophyte taxa that have been reported as having weedy characteristics within Australia, and 2) assign an origin status for these taxa, including those speculated in literature to be introduced by humans since European occupation.

Methods

A literature search was undertaken for Australian references to bryophyte taxa that are re-

ported as either introduced, adventive, or with weedy characteristics within all or part of their documented geographic range. Such information occurs across a diverse range of publications including taxonomic treatments, society newsletters, plant censuses and government reports. A list of taxa was compiled and then each taxon was provisionally assigned an origin status from majority agreement in literature or best estimate based on available information.

Categories were defined as:

- Introduced – speculatively or confidently determined as introduced from outside Australia since European occupation.
- Dual Origin – considered native to some parts of Australia but spread by humans, since European occupation, to also occupy other disjunct areas within Australia.
- Cosmopolitan – regarded as being distributed effectively worldwide and naturally occurring throughout their geographic range (with few exceptions). This may also include a bipolar temperate zone distribution.
- Cryptogenic – includes taxa that have been suggested as possibly introduced, with low certainty, and usually with unclear information for various reasons, e.g. insufficient data, but, where available, observations suggest possible introduction to Australia.

Invasive status was assigned to each taxon according to definitions in Richardson *et al.* (2000). Records of relevant taxa were retrieved from the Australasian Virtual Herbarium (Council of the Heads of Australasian Herbaria [CHAH] 2020a) to obtain a list of herbarium specimens that are geographically attributed to Australian states or territories. The collection year for each specimen was recorded and, where gaps appeared in data, the year was extracted from the 'Verbatim event date' field with verification of collector details. Records without year or collector information were removed. Year was then rounded down to decade, and frequency of records per decade was obtained for each taxon. A heatmap of collection frequency per decade was produced with ggplot2 (Wickham 2016) with data preparation using dplyr (Wickham *et al.* 2018) in R for Windows version 4.0.2 (R Core Team 2013).

Bryophyte nomenclature in Table 1 follows the Australian Plant Census—AusMoss (CHAH 2020b).

Assigning origin

For some introduced species, the evidence of their origin is compelling based on genetic analyses (Paterson *et al.* 2009; Carter 2010), horticultural records (Robinson 2003) or biogeographic analyses (Zenni 2013). There is less certainty as to whether some species occupied Australia before or arrived after European occupation. Some do not appear to occupy niches in natural systems, either because they are recent introductions that have not had time to disperse and naturalise, or are excluded by competition, soil conditions or some other limiting factor. A regional example from the vascular flora is provided – *Rytidosperma popinense* (D.I.Morris) A.M.Humphreys & H.P.Linder. Previously thought to be a Tasmanian endemic and threatened species, it was recently placed into synonymy with the more widespread *Rytidosperma fulvum* (Vickery) A.M.Humphreys & H.P.Linder, which is native to mainland Australia (Lorimer 2014). The Tasmanian entity was known only from roadsides and similar modified areas, leading to circumstantial arguments regarding its origin. There are similar circumstances for several bryophyte species that are associated with human activity and

found mostly around human population centres or high traffic areas. Such environments are suited to a number of ubiquitous species found in natural and modified areas, some of which have an apparent cosmopolitan distribution. One such species is the moss *Tortula muralis* Hedw. The first Australian collection was made in 1883 from a stone wall in Melbourne, and several other collections were made over the next 15 years from Sydney, Hobart, Perth and elsewhere (CHAH 2020a). Almost all these collections attribute the substrate as anthropogenic, including stone culverts, street walls, laneways and buildings. A collection from the Pyrenees Ranges in Victoria was made by Daniel Sullivan in 1884 (CHAH 2020a). This collection suggests a more remote and natural habitat, yet gold prospecting took place in the general area from about 30 years prior (Wilkie 2014). Such activities included extensive soil and vegetation disturbance. The origin of early Australian collections forms an argument for possible introduction of *T. muralis*. It could be speculated that *T. muralis* was introduced in rock ballast, imported bricks or similar materials. Determining origin can be more difficult when there are very few records to form a view of a species' ecological envelope. The moss *Ephemerum recurvifolium* has had its origin questioned in Australia (Fife 2014a) and has been recorded only from claypans along the Silver City Highway in NSW (Stone 2006; CHAH 2020a), where it is represented by a duplicated collection from one location. The recorded location is not precise, although many claypans occur in the general area, comprising a series of lakes along the Great Darling Anabranch that are subject to infrequent inundation (Murray Darling Basin Authority 2012). This species may be a recent European introduction to Australia, where it has become naturalised in habitats with similar characteristics to those in its northern hemisphere range (Novotný 1986). Alternatively, it may have been transported long ago (pre 1788) by birds (see Lewis *et al.* 2014) that migrated between Australia and other locations where *E. recurvifolium* occurs. In Australia, it is an introduced species under the first scenario, but it is likely a threatened species under the second scenario, thus determining

origin has clear implications for conservation management.

Taxa that are considered introduced to Australia since European occupation were probably imported by accident. Reproductive material is microscopic and can be imported in soil, on vascular plants and amongst various other imported goods (Dickson 1967; Ramsay *et al.* 1993). The moss *Syrrhopodon platycerii*, which occurs in Queensland and New South Wales, was reported to have been transported on one occasion from Queensland to Victoria on an ornamental fern (Meagher 2002). The cosmopolitan moss *Funaria hygrometrica* was recorded on sub-Antarctic Macquarie Island on a mound of imported builder's sand, although it has apparently not established elsewhere on the island (Seppelt 2004). In most other cases, there is no evidence such as this, to directly attribute the origin to a suspected anthropogenic introduction. Circumstances for Macquarie Island are atypical in that the island is geographically isolated and has a clearly definable area, which has been the subject of consistent survey since 1882 (Seppelt 2004). *Funaria hygrometrica* spores are capable of dispersing via the atmosphere (Glime 2017) although the sub-Antarctic/Antarctic is the only territory where this cosmopolitan moss does not appear to occur naturally within Australia's jurisdiction. When considering effective introduction, ecophysiological thresholds that limit establishment are of equal importance to dispersal capacity (Wikland and Rydin 2004), and processes that drive ecological filtering must be considered. *Sphagnum subnitens* Russow & Warnst. has several traits which have allowed it to spread successfully and become invasive in New Zealand within a short timeframe (~60 years). It is monoicous with gametophytes that are capable of self-fertilisation and without inbreeding depression; consequently, should only vegetative material be dispersed, a single gametophyte can establish and produce sporophytes. Establishment in New Zealand is reported from two likely anthropogenic introductions, based on the presence of two highly divergent genotypes. It has low genetic diversity throughout New Zealand populations despite its success in

various widely distributed habitats (Karlin *et al.* 2011). This species exemplifies trait combinations which make some species more successful than others at establishing in new areas.

Non-anthropogenic dispersal

While some plants can naturally disperse between continents, their typical dispersal distances are much shorter, e.g. within a landscape (Nathan 2006). A species may disperse from one continent to another more than once (Howarth *et al.* 2003) and, when considering timeframes of several thousand years or more, it is expected that such species will, on occasion, establish on a new continent. Long distance dispersal by spores and gametophyte material is reported in many bryophytes (Laaka-Lindberg *et al.* 2003; Lewis *et al.* 2014; Patiño and Vanderpoorten 2018). Gametophytes produce various vegetative propagules such as gemmae, deciduous shoots, bulbils, filamentous axillary rhizoids, leaf and/or stem fragments, all of which have the potential to develop into new gametophytes. Once arrived, success of establishment and the period that the species' invasion is incipient varies depending on suitability of its traits to a range of ecological conditions (Richardson *et al.* 2000). For example, ploidy level (Soltis and Soltis 2000) and climatic range tolerance (Gallagher *et al.* 2015) may contribute to how quickly new arrivals can establish and become invasive. When considering potential dispersal capacity and the persistence of bryophytes on earth for 400+ million years (Morris *et al.* 2018), it is tempting to question whether the origin of a species at a new location can be irrefutably resolved, at least without employing genetic analysis. Evidence offered for most species, however, is given in a similar light and with the same sorts of circumstantial arguments used for many vascular weeds, namely that they do not appear to occupy natural plant assemblages and that records are from relatively recent European occupation (Fensham and Laffineur 2019). Relying on collection records alone for evidence of origin has limitations (Meagher 2015; Patiño and Vanderpoorten 2015), although it can prove useful in conjunction with other corroborating evidence.



Fig. 1. *Pseudoscleropodium purum* with sporophytes from The Basin, Victoria.

Case study

The moss *Pseudoscleropodium purum* (Hedw.) M.Fleisch. (Fig. 1) is perhaps the most widely known invasive species in the Australian bryophyte flora and offers a good case study for examining concepts of bryophyte introductions. In 1976, Scott and Stone noted that it was not yet common in southern Australia. In the same year it was noted to be extremely widespread in at least one region of New Zealand (Fife 2020). Milne and Jolley (2010) reviewed its origins and weediness in Australia, regarding it as widespread in natural and disturbed habitats. It has since gained a high risk rating in Victoria for its combination of traits: having a typically significant impact on natural systems, having extensive potential for further spread, being highly invasive, having a rapid rate of dispersal, and affecting a restricted range of susceptible habitat types (White *et al.* 2018). Despite reports on its capacity to invade natural areas (Lepp 2015), the ecological implications in southern Australia are uncertain (Meagher and Fuhrer 2003; Seppelt *et al.* 2011). Its spread is effected primarily by passive dispersal of branch fragments

(Fritz 2009; Miller and Trigoboff 2001). Sporophytes are rare in Australia and New Zealand (Scott and Stone 1976; Milne and Jolley 2010; Seppelt 2020; Fife 2020) with only two collections bearing sporophytes in Australia, namely from The Basin in Victoria (Fig. 1) (pers. obs.) and within the Australian National Botanic Gardens in Canberra (CANB 653348.1). In a global review of bryophyte invasions, *P. purum* was the second highest ranking species for the number of regions which it had invaded (Essl *et al.* 2013). Herbarium records show that it was imported to Australia some time prior to 1946 (Milne and Jolley 2010; CHAH 2020a) and anthropogenic introduction to areas elsewhere in the world are reported from as early as 1875 (Dickson 1967). The first Australian collection in 1946 was from Cheltenham (Victoria) by botanist Trevor Clifford, who undertook various studies on bryophytes, and was noted as part of a resurgence in Australian bryological expertise at the time (Ramsay 2006). Another European moss in the same family (Brachytheciaceae), *Scleropodium touretii*, is reported in Australia from one location in Hobart, where

it co-occurs with, but occupies a different niche from, *P. purum* (Seppelt *et al.* 2011). This site, near Cascade Brewery, is also the location of the first observation of *P. purum* in Tasmania (Ratkowsky and Ratkowsky 1983).

Speculation that *P. purum* was probably introduced to Tasmania during the 1950s (Seppelt *et al.* 2011), is somewhat supported by an assessment of global bryophyte introductions, which indicates an average lag time of 40 years between detecting an introduction and detecting impacts (Essl *et al.* 2014). It was first recorded in Tasmania in the 1980s (Seppelt *et al.* 2011) although accurately estimating time of introduction is confounded by several factors. From the 1950s, other circumstances, such as the intensification of activities which assist dispersal, have included road network improvements (Wilson 2005). Also, the aforementioned resurgence in Australian bryological expertise may contribute to the temporal pattern of collections. Lepp (2012) noted that there was virtually no bryological activity (exploration/research) in Australia in the 1930s and 1940s (reflected in Fig. 2). According to genetic analysis, Australian *P. purum* material came from England and/

or France; the same sources appear in New Zealand and Canada (Fritz 2009). Introduction directly from England during earlier European settlement is plausible. Fritz (2009) speculated that *P. purum* was brought to Canada, Australia and New Zealand as packing material for trees, other living plant material, or with animal transports, as indicated elsewhere by other authors. Between 1812 and 1853, Hobart received numerous ships with over 74000 convicts and supplies, predominantly from England (Eldershaw 2003). These were followed closely (1840s–1860s) by a significant amount of livestock importation from Victoria via (now) Queens Domain and greater Hobart (Caldow 2010), which included transportation of plant material for feed (Caldow 2012). For decades to follow, shipping remained the primary means of importing and exporting goods more widely. In addition, the construction of cities, spread of agriculture, forestry and gold prospecting in southern Australia broadened opportunities for the spread and naturalisation of introduced species. Whether or not such early introductions of these species occurred, circumstances conducive to the importation and spread of



Fig. 2. Frequency of herbarium collections by decade for focus taxa within Australia (n=5049)

introduced species during the 19th century should not be disregarded. This requires a lengthy period of incipience or acclimatisation that may seem implausible based on usual expectations of weed invasions. Such expectations, however, are poorly grounded for bryophytes, as most observations about the timing and manner of suspected anthropogenic introduction are anecdotal. Similar considerations have been made by Seppelt and Cave (2011) about the manner of establishment for the moss *Eurhynchium praelongum* in Tasmania.

The current distribution of *P. purum* in Australia falls between southern Tasmania, Adelaide and Sydney. Its area of occupancy is 536 km² and extent of occurrence 653 386 km² (ALA 2020), which is not surprising, given its apparent ease of spread and 70+ years since it was introduced. Interpretation of its invasiveness and rate of spread requires consideration of the duration of its occupancy and current distribution. There has been no ecological research to date which quantifies invasiveness and ecological impacts of *P. purum* in Australia. Personal observations of occurrences in Victoria and Tasmania indicate association with at least some degree of prior vegetation and/or soil disturbance, even if disturbance has not occurred for decades. These observations include infestations in flood zones and along roadsides, deer tracks, walking tracks, pipelines, stock routes and similar areas where it has been dispersed by human activities (notably roadside slashing), animals or in flood water. The capacity for *P. purum* to spread is evident; however, without focused quantitative research, other factors controlling its recruitment and further spread in different types and qualities of native vegetation remain unresolved.

Other factors

Another factor to consider is that many species, both native and exotic, go unnoticed for an extended period prior to their discovery. This is sometimes the basis for new taxonomic discoveries. In the case of *Scleropodium touretii*, a lack of geographic context contributed to misinterpretation of the taxon as a novel endemic species, *Scleropodium australe* Hedenäs (Carter 2010). Consequently, its status as a new introduction went unnoticed. Another moss

regarded as introduced to Australia, *Barbula unguiculata* Hedw., was first reported in Tasmania in 1893 (Weymouth 1903), with at least some of Weymouth's material accessioned and described by Rodway (1914), and later re-examined with tentative confirmation by Sainsbury (1953). The species was not reported again from Tasmania until 1980, although it was collected elsewhere in Australia prior to this (Willis 1955a; CHAH 2020a). If these collections represent the same taxon, then there was a considerable period during which it was overlooked in Australia. Some species are evidently more puzzling in this regard than others, but as enigmatic species are increasingly documented, their recognition alone tends to raise interest in other observers to make new records. Further consideration should be given to the time that is taken to notice that an introduction has occurred, due to confusion with other species. Other confounding factors include the manner by which a species is introduced and its rarity within a region.

Taxa with weedy characteristics in Australia

There are 29 taxa (25 mosses and four liverworts) that are reported as weeds or at least as repeated casual introductions since European occupation in all or part of Australia (Table 1). Taxa listed in Table 1 should not be interpreted as a tally of anthropogenic introductions to Australia. Species listed are provisionally assigned a category of origin, with 13 taxa regarded as introduced since European occupation, six having dual origin, two having cosmopolitan distributions, and eight with uncertain (cryptogenic) origin. The number of Australia's introduced mosses and liverworts is comparable with that of New Zealand, where there are 14 introduced moss taxa (Manaaki Whenua Landcare Research 2020) and five introduced liverwort taxa reported (Engel and Glennly 2008). An additional 11 taxa are considered for their potential for introduction into Australia, either based on reports of their commercial uses, or their introduced status in New Zealand. The temporal distribution of bryophyte collections (Fig. 2) assists in understanding general patterns in the detection and reporting of suspected introduced taxa. Records indicate that cosmopolitan taxa that are widespread in

Table 1. Bryophyte taxa with published commentary of introduced or uncertain origin in Australia.

States and territories occupied according to distribution in CHAH (2020a) or references provided, are included below each name. LHI = Lord Howe Island, HIMI = Heard and McDonald Islands, Macq = Macquarie Island. The 'other sources' column includes other literature which states or speculates taxa to be introduced to Australia. Invasive status follows definitions in Richardson *et al.* (2000) where applicable.

Taxon/commentary	Origin	Invasive status	Other sources
<i>Aloina aloides</i> var. <i>ambigua</i> (Bruch & Schimp.) E.J.Craig SA, VIC, NSW Willis (1955b) as <i>Aloina ambigua</i> , at the time considered a new Australian record on a roadside bank near Marion, Adelaide SA. Catchside (1980) 'Probably an introduced species, in view of its occurrence on disturbed soils.'	Introduced	Naturalised	Glenny <i>et al.</i> (2011) note that this taxon was first recorded in New Zealand in 2007, where found as a ruderal species on roadsides.
Jolley and Milne (2012) 'Occurs on calcareous soil on roadsides and in disturbed habitats in south-eastern S.A., southern N.S.W., A.C.T. and Vic. Possibly introduced into Australia, this moss is native to Europe, North Africa and North America.'			
<i>Amblystegium serpens</i> (Hedw.) Bruch & Schimp. SA, TAS, VIC, NSW, QLD, HIMI, Macq Catchside (1980) 'On soil, stones and decaying wood in moist, shady places ... in situations such as conservatories, which suggest introduction.'	Dual origin	Naturalised	-
<i>Barbula unguiculata</i> Hedw. WA, SA, TAS, VIC, NSW, ACT, QLD Willis (1955a) 'abundant on damp shaded soil in a garden (Brighton Vic). A very common and variable species almost throughout the northern hemisphere. The habitat and lack of other Victorian occurrences would suggest that it is an introduction here.'	Introduced	Invasive	Downing <i>et al.</i> (2007) Scott and Stone (1976) Foreman <i>et al.</i> (2004) Meagher and Fuhrer (2003) White <i>et al.</i> (2018)
<i>Brachythecium albicans</i> (Hedw.) Schimp. SA, TAS, VIC, NSW, ACT, ?QLD Scott and Stone (1976) 'It seems certain that this species is introduced, originally from the northern hemisphere ... but it is now not uncommon in Vic and perhaps elsewhere.' Hedenäs (2012) 'Many collections of this species were made in anthropogenic habitats, and it had been assumed that the species was originally introduced in Australia ... however, since it also occurs in natural environments its status as introduced requires further investigation.'	Dual origin	Invasive	Foreman <i>et al.</i> (2004) Hedenäs (2002) Meagher and Fuhrer (2003) White <i>et al.</i> (2018) Willis (1955b)
<i>Brachythecium mildenium</i> (Schimp.) Milde SA, VIC, NSW, ACT Hedenäs (2012) 'All Australian specimens of <i>B. mildenium</i> have been collected in strongly anthropogenic habitats, pointing to its introduction since European settlement in the late 18th century. It can be found in similar habitats in Europe, but it also occurs in somewhat nutrient-rich fens and swampy forests.'	Introduced	Naturalised	-

Table 1. continued

Taxon/commentary	Origin	Invasive status	Other sources
<i>Brachythecium rutabulum</i> (Hedw.) Schimp. ?WA, SA, TAS, VIC, NSW, ACT, Macq. Catchside (1980) 'Fairly common in sheltered woodlands among grass and on the bases of trees and especially in lawns; perhaps introduced, at least to garden sites.'	Dual origin	Naturalised	-
<i>Bryum argenteum</i> Hedw. All states and territories Spence and Ramsay (2006) 'The abundance of <i>B. argenteum</i> in cities, in developed landscapes and other disturbed habitats, along with its general absence from native vegetation, suggest that it may have been introduced into Australia.'	Cryptogenic	-	Dalton (1998) Spence and Ramsay (2019)
<i>Calliergonella cuspidata</i> (Hedw.) Loeske TAS, VIC, NSW, ACT Klazenga (2012) 'Widespread in the temperate and boreal Northern Hemisphere; probably introduced into Australia and New Zealand.'	Introduced	Naturalised	-
<i>Ceratodon purpureus</i> (Hedw.) Brid. WA, SA, TAS, VIC, NSW, ACT, QLD, LHI, HIMI, Macq. Noted by Catchside and Stone (1988) as weedy and remarkably absent from NT.	Cosmopolitan	-	Dalton (1998)
<i>Chonecolea doellingeri</i> (Nees) Grolle NSW, QLD Ramsay et al. (1993) 'This species may have arrived as a propagule and become naturalised here.'	Cryptogenic	-	Brown and Coveny (1999) Engel and Glennly (2008)
<i>Ephemerum recurvifolium</i> (Dicks.) Boulay NSW Stone (2006) 'Occurs on soil crust with algae, lichens and other bryophytes in roadside claypans between Wentworth and Broken Hill in south-western N.S.W.; also in Europe, Russia and North Africa.' Fife (2014a) notes that despite Stone (1996) reporting this taxon as indigenous to Australia, its distribution reported elsewhere suggests it is adventive in Australia.	Cryptogenic	-	-

Table 1. continued

Taxon/commentary	Origin	Invasive status	Other sources
<i>Eurhynchium praelongum</i> (Hedw.) Bruch & Schimp SA, TAS, VIC, NSW Seppelt and Cave (2011) 'In the Southern Hemisphere the species is considered to be introduced in New Zealand (Sainsbury 1955), in southern Australia and Tasmania (Scott and Stone 1976), where it is believed to have at least in part been introduced.'	Invasive	Introduced	Catcheside (1980) Dalton (1998) Hedenäs (2012) Jones (2012) Meagher and Fuhrer (2003) Scott and Stone (1976) Sainsbury (1955) Catcheside (1980) Hedenäs (2012)
<i>Eurhynchium pumilum</i> (Wilson) Schimp. SA, VIC, NSW Hedenäs (2002) - 'Known only from one locality in South Australia, where it has been introduced; altitude unknown, on soil in shady part of a garden. Outside Australia occurring in Europe and northern Africa.' This taxon has been collected also from a creek bank in suburban Melbourne (MEL 2065360E) and from a nature/recreational reserve near Newcastle (NSW 755406).	Cryptogenic	-	
<i>Funaria hygrometrica</i> Hedw. All states and territories See notes in text. Fife (2019) 'Funaria hygrometrica is a serious greenhouse weed.' Fife and Seppelt (2012) 'Occurs in all States and Territories. Plants form patches or scattered stems on disturbed ground, especially the sites of fires, on burnt wood and old walls; very common in plant nurseries, on rubbish and apparently associated with high potash concentrations.'	Cosmopolitan (introduced Macquarie Island)	Casual for Macquarie Island	Seppelt (2004)
<i>Gemmabryum klinggraffii</i> (Schimp.) J.R.Spence & H.P.Ramsay WA, NT, NSW, LHI Spence and Ramsay (2019) 'A species of disturbed soil ... probably introduced.'	Introduced	Naturalised	Spence and Ramsay (2012)
<i>Gemmabryum tenuisetum</i> (Lampr.) J.R.Spence & H.P.Ramsay VIC, QLD Spence and Ramsay (2012, 2019) 'a rare scattered northern hemisphere disjunct species of disturbed soil ... probably introduced.'	Introduced	Naturalised	-
<i>Leptobryum pyriforme</i> (Hedw.) Wilson All states and territories Considered as introduced to Australia by Ramsay et al. (1993). Ramsay (2012) 'An annual, almost cosmopolitan species that is found on all continents except Antarctica; most common in temperate regions. This is a weed in pots in glasshouses; infrequent on damp earth, burnt soil or limestone.'	Cryptogenic	-	-

Table 1. continued

Taxon/commentary	Origin	Invasive status	Other sources
<i>Lunularia cruciata</i> (L.) Dumort. WA, SA, TAS, VIC, NSW, ACT, QLD, LHI Rowley (1916) 'Very common in glasshouses and gardens. Introduced' Meagher (2015) 'The thallose liverwort <i>Lunularia cruciata</i> (L.) Dumort. is a well-known coloniser of soil in disturbed damp or wet habitats, such as horticultural pots, gardens, roadsides and degraded stream banks. That it is introduced into Australia is beyond doubt ...'	Introduced	Invasive	Downing <i>et al.</i> (2007) Engel and Glennly (2008)
<i>Marchantia polymorpha</i> L. WA, SA, TAS, VIC, NSW, ACT, QLD, HIMI Meagher and Fuhrer (2003) 'An introduction from Europe.'	Introduced	Invasive	CANBR (2019a) Downing <i>et al.</i> (2007) Engel and Glennly (2008) White <i>et al.</i> (2018)
<i>Pseudoscleropodium purum</i> (Hedw.) M.Fleisch. SA, TAS, VIC, NSW, ACT See discussion in text including additional references. Forman <i>et al.</i> (2003) Meagher and Fuhrer (2003) Milne and Jolley (2010)	Introduced	Invasive	CANBR (2019b)
<i>Rhytidadelphus squarrosus</i> (Hedw.) Warnst. TAS Reported by Dalton (1997) as introduced to Tasmania. It occupies open grassy areas including lawns and golf courses. Now recorded at numerous sites in western and southern Tasmania.	Introduced	Invasive	Dalton (1998) Dalton <i>et al.</i> (2015) Fife (2014b) Lepp (2015)
<i>Rosulabryum rubens</i> (Hedwig) J.R.Spence VIC, NSW, QLD Spence and Ramsay (2006) 'Known from disturbed soil in QLD, N.S.W. and Vic. Also in Europe, North America, India, Japan, Malaysia and New Zealand. It is often found on disturbed soils, and it may have been introduced' Spence and Ramsay (2019) - 'This species is local in Australasia, on disturbed soil and rock, often concrete, and is probably introduced.'	Introduced	Naturalised	Ramsay <i>et al.</i> (1993)
<i>Rosulabryum tuberosum</i> (Mohamed & Damanhuri) J.R.Spence QLD Spence and Ramsay (2006) 'Known from two localities in north-eastern Qld; possibly introduced. Also in Malaysia and New Guinea.' (J.R.Spence, unpublished data)	Cryptogenic	-	-
<i>Sphaerocarpos texanus</i> Austin SA, VIC, NSW, ACT Scott (1985) 'Perhaps it is introduced and spreading [in southern Aust.]' Meagher and Fuhrer (2003) '... one of the few liverworts in Australia that seems certain to have been introduced.'	Introduced	Naturalised	CANBR (2019a) Engel and Glennly (2008)

Table 1. continued

Taxon/commentary	Origin	Invasive status	Other sources
<i>Syntrichia papillosa</i> (Wilson) Jnr. WA, SA, TAS, VIC, NSW, ACT, QLD, LHI Dalton (1998) '... <i>Tortula papillosa</i> ... is found as an epiphyte on introduced trees in parklands throughout the State [Tas] and as yet hasn't established itself on phorophytes in native vegetation.' Rodway (1914) 'Very common on elm, willow etc. in gardens.'	Dual origin	Naturalised	-
<i>Syntrichia pygmaea</i> (Dusén) R.H.Zander VIC, NSW Reported new to Australia by Solliman (1995) based on a collection from a historic hotel site near Dargo, Vic. Another collection from coastal NSW pre-dates the former. Both specimens are associated with developed sites, the one at Dargo on bark of an exotic tree.	Cryptogenic	-	CHAH (2020a)
<i>Tortula muralis</i> Hedw. WA, SA, TAS, VIC NSW, ACT, QLD Scott and Stone (1976) 'Throughout the world, although it is now virtually impossible to tell where it is native and where introduced.' See further discussion in text.	Cryptogenic, probably cosmopolitan	-	-
<i>Tortula truncata</i> (Hedw.) Mitt WA, SA, TAS, VIC, NSW, ACT, QLD Scott and Stone (1976) '... growing on bare soil on waste ground and in suburban gardens' Catchside (1980) 'Fairly common in South Australia ... in gardens and urban habitats, probably not native.'	Dual origin	Naturalised	White <i>et al.</i> (2018)
<i>Trichostomum brachydontium</i> Bruch. WA, SA, TAS, VIC, NSW, ACT, QLD, NT Reported as introduced to Australia by Downing <i>et al.</i> (2007). Possibly introduced to Tasmania where represented by a single record (collection)	Dual origin	Naturalised	CHAH (2020a)
Additional taxa for consideration <i>Fissidens bryioides</i> Hedw. Regarded as adventive in New Zealand due to association with anthropogenic habitats (Beever 2014). A variable species in the broad sense, recorded from Australia (QLD) in rainforest (Seppelt and Stone 2016). There are similar considerations for <i>Fissidens curvatus</i> Hornsch. in New Zealand where it is considered to have dual origin status (Beever 2014).	-	-	-

Table 1. concluded

Taxon/commentary	Origin	Invasive status	Other sources
<i>Fissidens dubius</i> P.Beauv. Recently introduced to New Zealand (Beever 2014). Not reported in Australia.	-	-	-
<i>Fissidens exilis</i> Hedw. Adventive in New Zealand (Beever 2014). Not reported in Australia.	-	-	-
<i>Fissidens taxifolius</i> Hedw. Adventive in New Zealand (Beever 2014). Not reported in Australia.	-	-	Beever (1996)
<i>Monosolenium tenerum</i> Griff. Thallose liverwort available in aquarium trade.	-	-	Gradstein (2003)
<i>Rhytidiadelphus triquetrus</i> (Hedw.) Warnst. Adventive in New Zealand. Fife (2014b) reports <i>Rhytidiadelphus triquetrus</i> as adventive in Tasmania. There does not appear to be a specimen attributed to this and there are no local publications on its presence in Australia.	-	-	-
<i>Riccia bullosa</i> Link Adventive in New Zealand (Engel and Glenny 2008).	-	-	-
<i>Riccia ciliata</i> Hoffm. Adventive in New Zealand (Engel and Glenny 2008).	-	-	Campbell (1977)
<i>Riccia glauca</i> L. Adventive in New Zealand (Engel and Glenny 2008).	-	-	Campbell (1977)
<i>Sphagnum subnitens</i> Russow & Warnst. Adventive in New Zealand (Engel and Glenny 2008).	-	-	-
<i>Vesicularia ferriei</i> (Cardot & Thér.) Broth. Nearest natural range is Philippines, where rare (Linis and Tan 2013). Used in aquarium trade.	-	-	-

Australia have a relatively consistent record since the 1800s, although all taxa have an increased frequency of collection since the 1940s. Those lacking or with few pre-1940 collections may support other circumstantial arguments for being 20th-century introductions.

Invasive bryophyte control

There are very few tested methods for managing ecological impacts caused by weedy bryophytes. Prior to attempting control, it is important to determine what the impacts are and how tractable the threat is. In most situations, chemical control may not be effective at removing an infestation while simultaneously causing off-target damage to other species. Zimmer *et al.* (2012) consider control of *P. purum* impractical without hand/mechanical removal. The use of glyphosate has been proposed on localised patches of *P. purum* and *B. albicans* (Couper 2003; Foreman *et al.* 2004); however, this herbicide has not been tested for such use. Commercial products for removing moss in lawns or golf greens include active ingredients that may have adverse impacts on native vegetation. Burning isolated patches of infestations may be effective in some situations.

Preventing the spread of propagules and avoiding soil disturbance in native vegetation is perhaps the best control measure for weedy bryophytes, in the absence of further information on this matter. Preventing the spread of propagules would involve, for example, cleaning machinery prior to transport and controlling livestock access. Most species in Table 1 have no reported ecological impacts, and reports of those that appear to impact natural areas are supported only by anecdotal observations. Experimental research on the impacts of a small number of introduced bryophyte species is warranted to determine the need and methods for control. It can be speculated that the ecological impacts of introduced bryophytes may at least include competition with native bryophytes and changes to vascular plant recruitment. In some cases, these effects may impact on the persistence of threatened species, with implications for conservation management.

Conclusion

The list of taxa in Table 1 serves as a provisional list of bryophytes with notable adventitious distributions of anthropogenic origin, within at least part of their Australian range. Some of these have weedy traits with the potential to be invasive in native vegetation, particularly that which has been subject to historical and recent clearing. There are, no doubt, other taxa that are suspected to be in the same class of bryophytes. Chronological analysis of herbarium records alone is confounded by various factors, including consistency in survey effort and variation in the establishment mechanisms of different taxa. It is expected that the origins of cryptogenic taxa may be resolved in time with genetic analysis. Ecological research is required to gain a better understanding of threats to native habitats, caused by a small number of invasive bryophyte taxa.

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Fifty-two Years Ago

Two Botanical Notes

By JB CLELAND

Blackberries

Baron von Mueller, whose era in Victoria extended from 1852 till his death in 1896, distributed the seeds of blackberry (**Rubus fruticosus* agg.) during his botanical expeditions partly to prevent erosion (see *Vict. Nat.* 76, June 1959, p. 33; 77, Jan, 1961, p. 258; and 77, April 1961, p. 354). The Baron was in South Australia before he went to Victoria but it seems unlikely that he distributed any blackberry seeds in South Australia, although some representative of the *Rubus fruticosus* group was no doubt deliberately introduced into this State during the early days.

Black also records the distinctive cut-leaf species **R. laciniatus*. It is quite likely that seeds of both were brought out by some emigrant who thought that one of these might succeed in establishing itself better than the other. If they were introduced together, and not on separate occasions, the first-named has far outstripped the other. It has reached, or been deliberately conveyed to, the Encounter Bay district but **R. laciniatus* has not yet been found there. Blackberry has been noted on Kangaroo Island. Eichler in his *Supplement* records four other species. If these are valid, this would mean that in South Australia there had been no less than six (or more) separate introductions of the blackberry deliberately selected. This is surely unlikely.

It may be pointed out that Australia lends itself to the study of variations in plants derived from a single source. In some instances the introduction has been so unlikely that it is improbable that it took place more than once. In such a case, variations in the descendants would be within the range of a species which may be particularly liable to throw minor mutants. Does this explain the occurrence of the variations which Eichler gives as species? It would be difficult to come to a sound conclusion in such a matter in Europe, but more simple in Australia.

The same reasoning applies to the introduced **Centaurium*, though seeds of this plant might readily have been accidentally brought here more than once. Eichler considers three species have arrived here in addition to the native *C. spicatum*. Black, under the name **Erythraea Centaurium*, wisely says "a variable species" and mentions under this name that some botanists separate **pulchellum* and **littoralis* as species. Under *C. minus* (without differentiation), the plant is recorded from both Encounter Bay and Kangaroo Island.

From *The Victorian Naturalist* 85, p. 252, September 5, 1968

The framework for the Victorian Government's management of weeds and pest animals on public land, for the protection of the state's biodiversity

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Abstract

Weeds and pest animals can be found in all terrestrial and aquatic habitats in Victoria, and have a significant impact on the environment, economy and the community. The Victorian Government operates within a framework guiding the management of weeds and pest animals to reduce their impact on values of importance to all Victorians, both in its role as the land manager of Victoria's state forest and protected areas (public land) and as a conservator of the state's biodiversity—a responsibility shared by all land managers. The framework is designed to be flexible to deliver cost-effective actions to counter existing and potential pest species. This paper focuses on the current (2020) framework for the Victorian Government's management of weeds and pest animals on public land, for the protection of the state's biodiversity. (*The Victorian Naturalist* 137(6), 2020, 195–202)

Keywords: weeds, pest animals, Victorian pest management framework, biodiversity conservation, public land

Introduction

Weeds and pest animals can be found in all terrestrial and aquatic habitats in Victoria, ranging from widespread species such as Blackberry *Rubus fruticosus*, the European Rabbit *Oryctolagus cuniculus* and the European Red Fox *Vulpes vulpes*, which have reached their likely extent, to species such as Buffel Grass *Cenchrus ciliaris*, Feral Goat *Capra hircus* and Feral Pig *Sus scrofa*, which are still expanding into new areas (DELWP 2017). Weeds and pest animals have a significant impact on the state's biodiversity, agricultural and social values, and their management is critical to the environment, economy, and community health and wellbeing. Weeds and pest animals cost Victoria over \$900 million a year in management and control programs (Commissioner for Environmental Sustainability 2008), and the introduction and spread of new pests continue to threaten Victoria's biodiversity and require vigilance and effective response. The Victorian Government operates within a framework guiding the management of weeds and pest animals to reduce their impact on values of importance to all Victorians, both in its role as the land manager of Victoria's state forest and protected areas (public land) and as a conser-

vator of the state's biodiversity—a responsibility shared by all land managers.

The management of weeds and pest animals in Victoria is guided by a framework with the necessary scope and flexibility to remain effective across a range of pest management objectives. The framework in its entirety is complex, directing and guiding the management of introduced weeds and pest animals to reduce their impact on agriculture, the economy, the environment, and community values, including human health and safety. Roles and responsibilities for pest management in Victoria are shared between a diverse range of stakeholders across government and non-government land managers, including Victoria's Traditional Owners (Aboriginal people), and the wider community. Given the large and diverse number of organisations and people invested in managing pests in Victoria, all with an equally large and diverse number of reasons for doing so, it is no surprise that the pest management system in its entirety is difficult to map. This paper will focus on the current (2020) framework for the Victorian Government's management of weeds and pest animals on public land, for the protection of the state's biodiversity (Fig. 1).

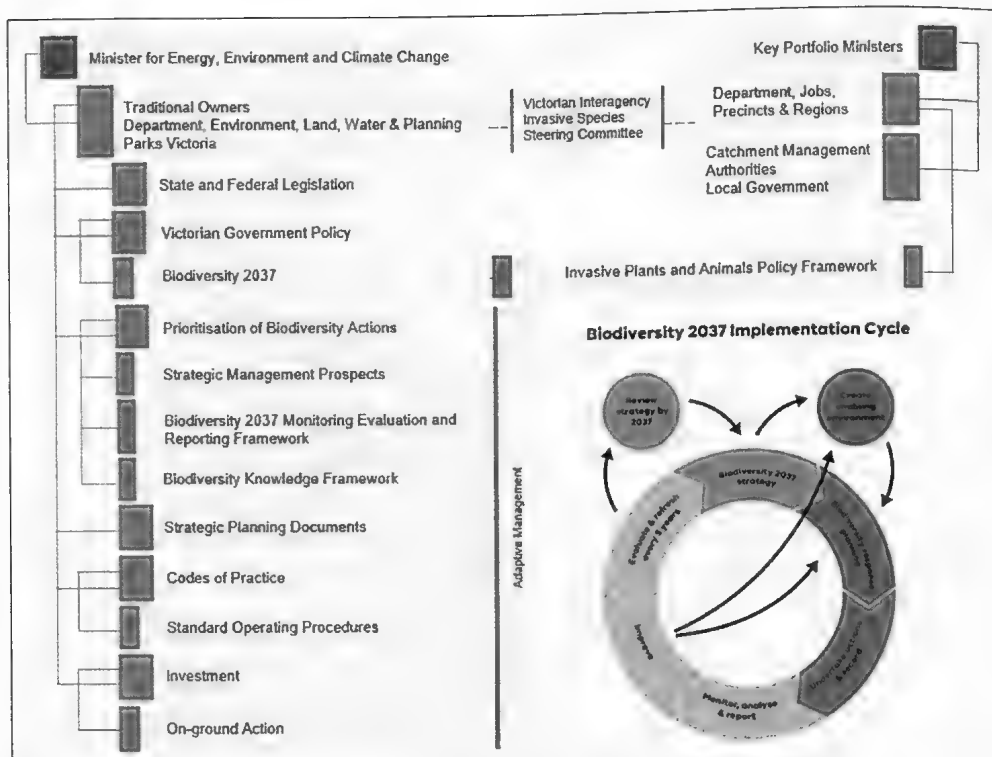


Fig. 1. The framework for the Victorian Government's management of weeds and pest animals on public land for the protection of the state's biodiversity.

Roles and responsibilities

The Minister for Energy, Environment and Climate Change is responsible for administration of the legislation that governs the management of the state's forests and protected areas (public land) and the protection of Victoria's biodiversity. The Secretary, Department of Environment, Land, Water and Planning (DELWP) and Parks Victoria have responsibility as the land manager for the management and control of weeds and pest animals on public land, with Traditional Owners playing an important role in joint management. The Department of Jobs, Precincts and Regions (DJPR) administers the state-wide invasive species policy, known as the Invasive Species Policy Framework. The Victorian Invasive Species Inter-Agency Steering Committee coordinates pest management between DELWP, Parks Victoria and DJPR. Biodiversity conservation and protection is the responsibility of all Victorian land managers, public and private.

Legislation

The management of weeds and pest animals in Victoria needs to be considered within a comprehensive legislative framework that incorporates both national and state responsibilities. Table 1 provides a summary of key relevant legislation, spanning obligations to control weeds and pest animals as set out in the *Catchment and Land Protection Act 1994*, through to requirements to protect native flora and fauna of national and state significance enshrined in the *Environment Protection and Biodiversity Conservation Act 1999* and *Flora and Fauna Guarantee Act 1988*.

Policy

Protecting Victoria's Environment—Biodiversity 2037 combined with the *Invasive Plants and Animals Policy Framework* (Agriculture Victoria 2010) provide the policy framework for the management of weeds and pest animals in Victoria for the protection of the state's biodiversity.

Table 1. Key State and Federal legislation, agreements and conventions relevant to the management of weeds and pest animals on public land in Victoria for the protection of the state's biodiversity.

Jurisdiction	Legislation	Description
National	<i>Environment Protection and Biodiversity Conservation Act 1999</i> (EPBC Act)	The EPBC Act is the Australian Government's central piece of environmental legislation, providing for the recognition of weeds and pest animals as threats to native animals and plants of national significance. Once a threat is listed under the EPBC Act, a threat abatement plan can be put into place. All jurisdictions, including Victoria, provide input into the development of national threat abatement plans.
Victoria	<i>Flora and Fauna Guarantee Act 1988</i> (FFG Act)	The FFG Act is the primary piece of Victorian legislation for the conservation of threatened species and ecological communities and the management of processes threatening Victoria's native flora and fauna. Under the FFG Act, such key threats as predation by the European Red Fox can be listed as a potentially threatening process and an action statement prepared.
National	<i>National Parks Act 1975</i> (NP Act)	The NP Act makes provisions in respect to national parks, state parks, marine national parks and marine sanctuaries. It requires the relevant Minister (as the land manager) to undertake weed and pest animal management in national and marine parks to preserve and protect indigenous flora and fauna.
Victoria	<i>Catchment and Land Protection Act 1994</i> (CaLP Act)	The CaLP Act is the key piece of legislation that regulates the control, keeping, movement, trade and release of noxious weeds and pest animals in Victoria. It provides for obligations on public and private land managers.
Victoria	<i>Prevention of Cruelty to Animals Act 1988</i> (POCTA)	POCTA sets out animal welfare requirements and places conditions on the use of control measures (e.g. trapping) for the management of pest animals.
National	<i>Agricultural and Veterinary Chemicals (Control of Use) Act 1992</i>	Outlines the requirements for the control and use of chemicals and poisons. This relates to the safe and appropriate use of vertebrate poisons such as sodium fluoroacetate (1080) or 4-amino propiophenone (PAPP).

Protecting Victoria's Environment—Biodiversity 2037

The Victorian Government released Victoria's key biodiversity policy, *Protecting Victoria's Environment—Biodiversity 2037* (Biodiversity 2037), in 2017, setting out the 20-year plan to stop the decline of Victoria's native plants and animals. Biodiversity 2037 recognises weeds and pest animals as one of the most significant drivers of the decline in the state's biodiversity, noting that between one quarter and one third of all of Victoria's terrestrial plants, birds, rep-

tiles, amphibians and mammals, along with numerous invertebrates and ecological communities, are considered threatened with extinction (DELWP 2017). Biodiversity 2037 has set targets for the management of weeds and pest animals in key locations as a priority action to help ensure Victoria's natural environment is healthy.

The Invasive Plants and Animals Policy Framework

Victoria's *Invasive Plants and Animals Policy Framework* (IPAPF) sets out the approach, or

overarching principles, that underpin weed and pest management in Victoria. The IPAPF adopts a risk management approach (also known as a 'biosecurity approach') that links policy outcomes to stages of invasion. The generalised invasion curve (Fig. 2) provides an indication of the likely economic returns from action taken to counter invasive species at different stages of invasion, demonstrating that prevention and eradication, stopping a weed or pest animal becoming established in the first place, is the most cost-effective course of action.

The risk management approach is based on four key elements: prevention, eradication, containment and asset based protection.

Prevention: Preventing the entry and establishment of new pest species in Victoria is considered the most cost-effective approach. While the Australian Government is responsible for managing national pre-border and border biosecurity, all states and territories work together through inter-governmental agreements to manage the risk of invasive species spreading across jurisdictional boundaries. Within Victoria, monitoring and surveillance to confirm absence, risk and pathway analysis are important tools to ensure that no new high-risk invasive species are introduced into the state.

Eradication: Early detection and rapid response is vital to ensure any high-risk new weed or pest animal that may cross the state border does not become established. Effective surveillance, monitoring and reporting processes are essential.

Containment: Containing the distribution of a weed or pest animal to a defined containment line and/or limiting the density of pest species that are beyond eradication but are not yet widespread.

Asset based protection: Prioritising the protection of highest priority values from the impacts of widespread weeds and pest animals. Asset based protection is the primary focus of public land managers when managing pests for the greatest benefit to biodiversity.

Biodiversity asset protection—identification and prioritisation of biodiversity actions

Widespread weeds and pest animals are found in all terrestrial and aquatic habitats in Victoria, making the process of prioritising where and how to act for the greatest benefit to biodiversity critically important. This is an important consideration for any government that seeks to optimise public good by delivering the most

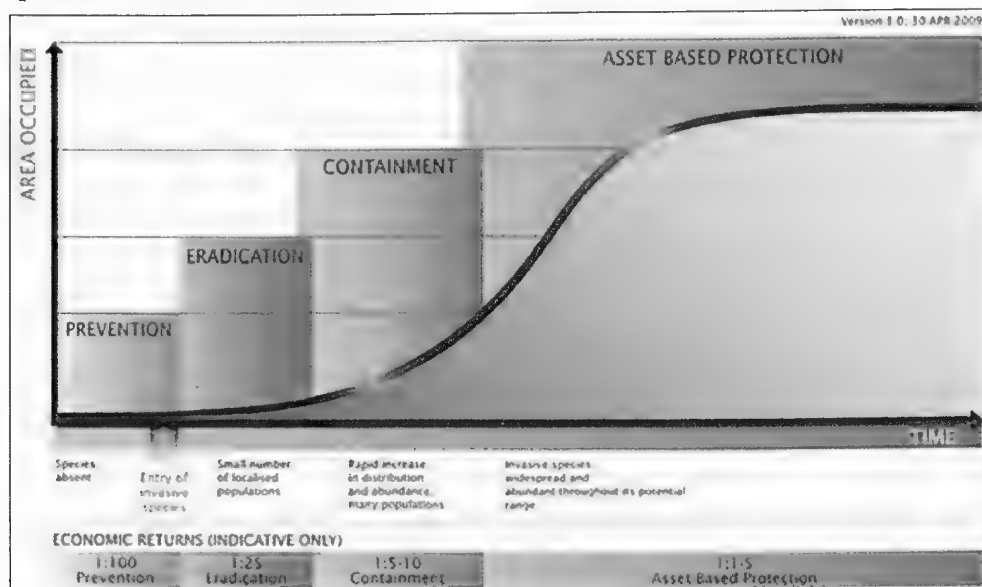


Fig. 2. Generalised invasion curve showing actions linked to each stage of invasion. Reproduced from Invasive Plants and Animals Policy Framework (Agriculture Victoria 2010), with permission.

cost-effective outcomes. Evidence-based decision-making is critical to improving outcomes for biodiversity. To help protect biodiversity from key threats such as weeds and pest animals, decision-support tools are necessary to deal with the complexity of nature in a way that is accessible to managers and stakeholders (Bruce *et al.* 2020; Thomson *et al.* 2020).

Over the past few years, DELWP has invested in the development of several decision-support tools to help land managers prioritise effort. Key decision-support tools such as Strategic Biodiversity Values, Habitat Distribution Models, Threat Models, Benefit of Action Models, and Strategic Management Prospects allow practitioners to quantify the benefits, identify cost-effective conservation actions, and better manage uncertainties. These tools can be viewed using NatureKit, DELWP's online biodiversity mapping and reporting platform (DELWP 2020a).

Strategic Management Prospects (SMP)

SMP, a key decision-support tool, uses spatially explicit models to integrate and simultaneously compare information on biodiversity values, threats, effectiveness of management actions and indicative costs of management actions for biodiversity across Victoria (Fig. 3). The result is a series of products that identify the most cost-effective conservation actions (including pest and weed management) that have the greatest benefit to the most species across Victoria (Thomson *et al.* 2020). SMP covers all land tenures, >4000 native species and 17 actions, and is a powerful decision-support tool for government and non-government biodiversity practitioners and conservation managers (Thomson *et al.* 2020).

Public land managers use SMP to help direct where efforts should be concentrated to control weeds or pest animals for the greatest benefit to biodiversity. SMP provides a method for land

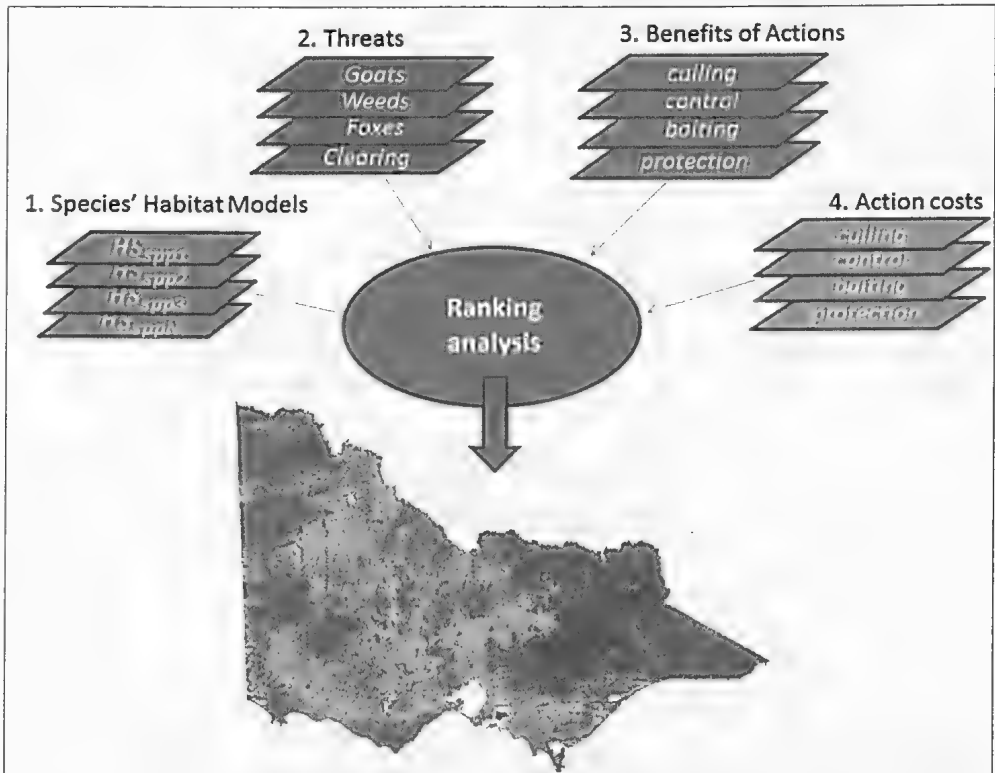


Fig. 3. Strategic Management Prospects—inputs to the analysis, spatial ranking process, and example outputs (DELWP 2018).

managers to compare and select management options. Figure 4 illustrates an example of an output from the SMP tool where the question asked was: where does integrated European Red Fox and Feral Cat control produce the greatest benefit for biodiversity in Victoria?

Figure 5 provides another example of how SMP has been used to prioritise weed and pest animal control following the 2019/2020 bushfires in East Gippsland and North East Victoria. DELWP's decision-support tools, including SMP, were instrumental in preparing *Victoria's Bushfire Emergency: Biodiversity Response and Recovery Report* (DELWP 2020b), which identified pest management as a high priority immediate action to protect species of concern. Figure 5 shows the indicative ranking of areas in East Gippsland for implementing European Red Fox control post-fire according to relative cost-effectiveness.

Monitoring, evaluation and reporting, and knowledge acquisition

Decision-support tools are used alongside a monitoring, evaluation and reporting, and knowledge framework. *Biodiversity 2037 Monitoring, Evaluation, Reporting and Improvements Framework (MERF) Version 2.0* (DELWP 2019a) is designed to demonstrate the progress of collective effort to deliver outcomes and targets and to embed continuous improvement into planning and action. Public land managers collect, analyse and report on targeted data, such as standard outputs and management effectiveness measures in support of the MERF and as an important driver for adaptive management. Adaptive management ensures public land managers utilise the results of their management interventions to test and refine their approach. The *Biodiversity Knowledge Framework, Version 1* (DELWP 2019b) describes DELWP's approach to identifying and filling

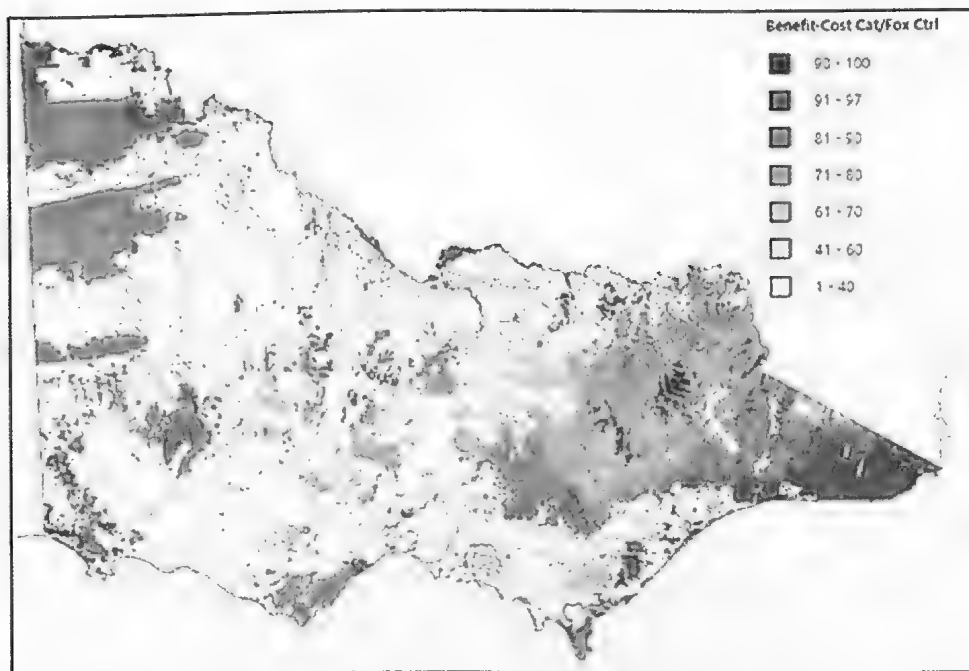


Fig. 4. Example of a Strategic Management Prospects 2.0 map showing the indicative ranking (or prioritisation) of areas for integrated European Red Fox and Feral Cat control according to relative cost-effectiveness. The darker blue indicates areas of higher ranking in terms of cost-effectiveness. Parts of Victoria are blank because Feral Cat control is not permitted in these areas under the CaLP Act. Map output produced from NatureKit.

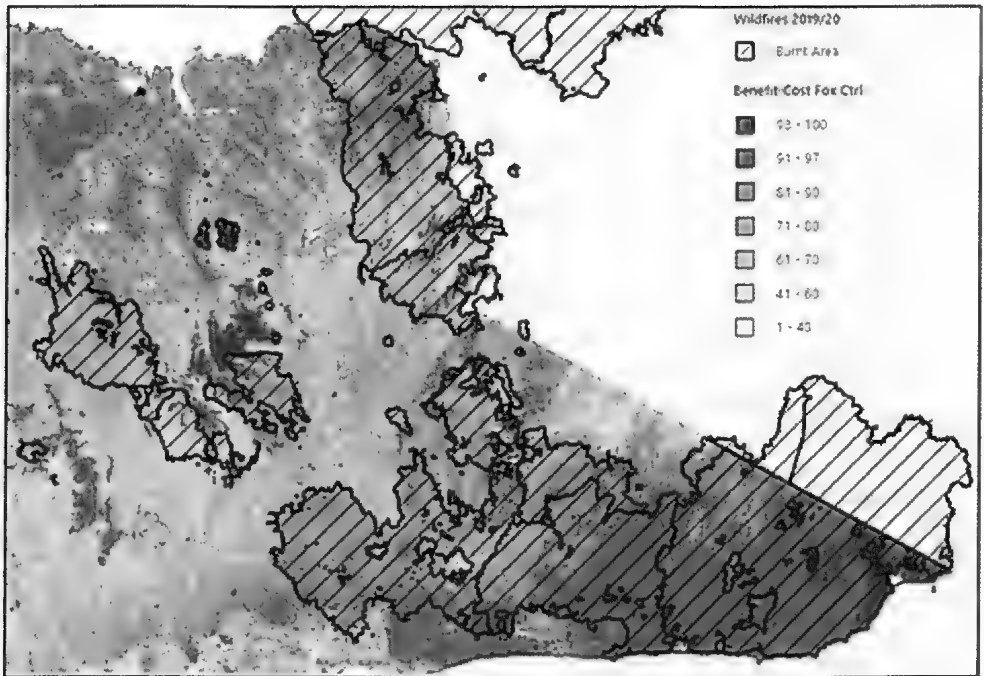


Fig. 5. Indicative ranking of areas for implementing European Red Fox control according to relative cost-effectiveness. High values (100) indicate areas, where Fox control has a relatively higher return on investment compared with low values (0). Dashed area shows the extent of the 2019/2020 wildfires (DELWP 2020b).

key biodiversity knowledge gaps to prioritise the acquisition of scientific knowledge, given limited resources and vast uncertainty.

Strategic planning and on-ground action

On-ground action to manage weeds and pest animals on public land sits within the context of a range of strategic planning documents. These include Regional Catchment Strategies, Parks Management Plans and Conservation Action Plans. DELWP and Parks Victoria plan and deliver on-ground pest management according to a Code of Practice Framework (Fig. 6) that provides information and guidance to public land managers on the best-practice management of specific pests. Adherence with the Code of Practice, including Standard Operating Procedures, is required for all state government staff and their contractors.

On-ground action

The Victorian Government is committed to reducing the impact of weeds and pest animals

on public land for the protection of the state's precious biodiversity. In 2019/2020, DELWP and Parks Victoria delivered targeted weed and pest animal management aimed at achieving enduring, cost-effective, meaningful biodiversity outcomes on over 2.2 million hectares, or one quarter of public land. The principles for government investment in weed and pest animal control on public land to achieve biodiversity protection outcomes have been developed, tested and refined over the past fifteen years, and have given rise to landscape-scale, collaborative management programs that operate at the optimal spatial and temporal scale to secure the greatest benefit for biodiversity.

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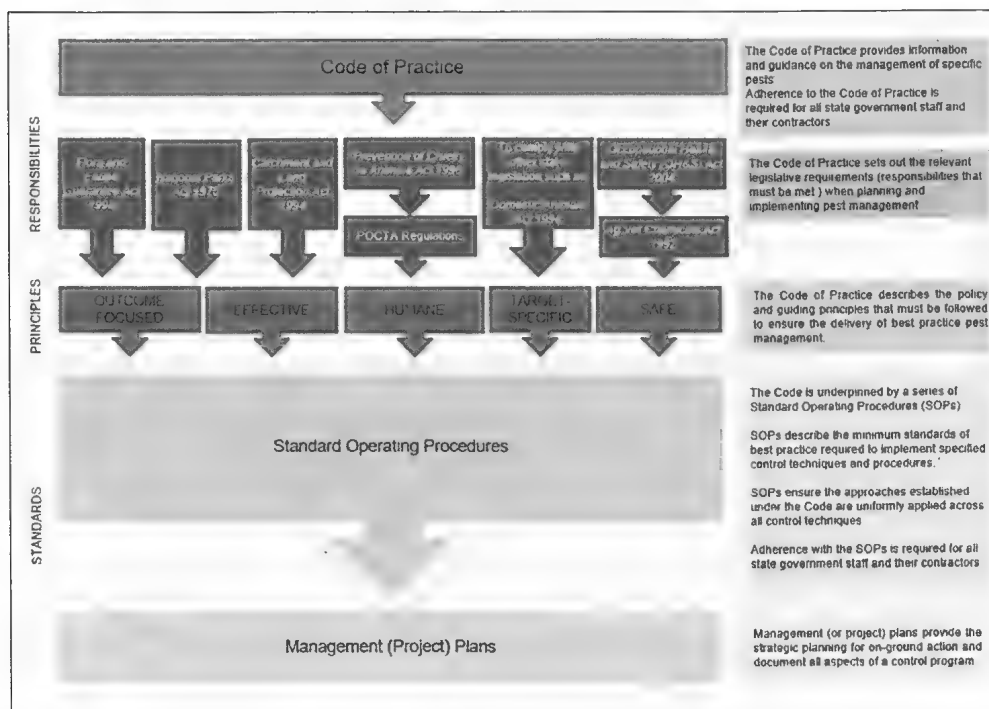


Fig. 6. Code of Practice Framework guiding pest management by state government staff and contractors on public land in Victoria.

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A policy approach to non-indigenous bird management in Victoria: managing potential threats to biodiversity, social amenity and economic values

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Abstract

Managing the potential adverse effects of invasive species on the economy, environment and social amenity requires leadership. Often this comes from government in the form of policy or other regulatory instruments at its disposal. Prior to 2014, there was a gap in the Victorian Government's ability to manage non-indigenous birds. Recognising this, we designed and implemented a reform process which consisted of a policy and species declaration under the *Catchment and Land Protection Act 1994*. Central to the policy development was stakeholder engagement. Here, we describe the construction and implementation of that policy. (*The Victorian Naturalist* 137(6), 2020, 203–209)

Keywords: alien species, invasions, regulation, pest birds, stakeholder engagement

Introduction

Alien or non-indigenous species' introductions and invasions are recognised as one of five global pressures on biodiversity, along with climate change, habitat change and degradation, over-exploitation, and pollution (Pereira *et al.* 2012; Blackburn *et al.* 2014; Bellard *et al.* 2016). Amongst vertebrates, birds are very successful invaders, often causing significant environmental and/or economic impacts (Duncan *et al.* 2003; Evans *et al.* 2016). The total economic loss resulting from the impacts of alien birds in just six countries (Australia, Brazil, India, South Africa, UK, and USA) was previously estimated to be greater than AU\$2.5 billion per year (Pimentel 2002), while three bird species (Common Starling *Sturnus vulgaris*, Common Myna *Acridotheres tristis* and Red-vented Bulbul *Pycnonotus cafer*) were included on the IUCN list of 100 of the world's worst invaders (Lowe *et al.* 2004).

In Australia, non-indigenous birds, as defined by Duncan *et al.* (2003), tend not to attract the same emotive 'call to arms' action when compared to other invasive alien vertebrates such as deer (e.g. Fallow Deer *Dama dama*, Red Deer

Cervus elaphus and Rusa Deer *Rusa timorensis*), rodents (e.g. House Mouse *Mus musculus* or Black Rat *Rattus rattus*), the European Rabbit *Oryctolagus cuniculus* or the European Red Fox *Vulpes vulpes*, yet their impacts are broad and wide reaching (Evans *et al.* 2014). Community perceptions of the value of wildlife are complicated (Ainsworth *et al.* 2018). This is particularly so for invasive species and birds in general. Until 2014 there was no clear mechanism to direct the management of non-indigenous birds or address the risks they pose in Victoria. Here, we describe the construction and implementation of a policy that has enabled management of non-indigenous birds in Victoria.

Non-indigenous birds in Victoria

There are nearly 250 non-indigenous species of birds housed in public and private aviculture and collections in Australia (Department of Agriculture, Water and the Environment 2007), with c. 200 of these bred and traded by aviculturists (Vall-Ilosera and Cassey 2017a). In addition, at least 20 alien species have established widespread populations on the Australian mainland (Olsen *et al.* 2006). Of these

non-indigenous bird species, 15 are known to be at large in Victoria.

The most notable of these are the Common Myna, Common Starling, European Blackbird *Turdus merula*, and House Sparrow *Passer domesticus*. These species are well-known to have negative impacts on commercial enterprises, native species and the wider environment as well as social amenity. For example, Grarock *et al.* (2012) found that the Common Myna had a negative impact on the long-term abundance of some cavity-nesting bird species (e.g. Sulphur-crested Cockatoo *Cacatua galerita*, Crimson Rosella *Platycercus elegans*, Laughing Kookaburra *Dacelo novaeguineae*) and some small bird species, including Superb Fairy-wren *Malurus cyaneus*, Striated Pardalote *Pardalotus striatus*, Willie Wagtail *Rhipidura leucophrys*, Grey Fantail *Rhipidura albiscapa*, Magpie-lark *Grallina cyanoleuca*, and Silvereye *Zosterops lateralis*. Similar displacement of native species by Common Starlings has been observed by Ingold (1998) and Koenig (2003). Non-indigenous birds are ubiquitous in the backyards of much of Victoria, yet few members of the general public lament the diversity of birdlife that could be present if these species were absent.

In addition to these widely established non-indigenous species, there are other bird species that often appear in the wild in small numbers or as occasional escapees, including Red-whiskered Bulbuls *Pycnonotus jocosus*, Barbary Doves *Streptopelia risoria* and Rose-ringed Parakeet *Psittacula krameri*. Propagule pressure and the global history of bird introductions suggests that, without intervention, it is a matter of 'when' rather than 'if' birds like this will become established (Cassey *et al.* 2004; Blackburn *et al.* 2015; Vall-lloera and Cassey 2017b). Prevention, early detection and rapid response to new incursions are the most cost-effective forms of invasive species management. This requires pre-emptive evidence policy approaches.

The role of government

Everyone has a responsibility for biosecurity, but there is a clearly defined leadership role for government to manage potential adverse effects of invasive species on the economy, environment and social amenity. Non-indigenous birds

have a wide range of potential and realised impacts in Victoria, particularly loss of horticulture production, reduced amenity value and threats to biodiversity. Having a clear policy approach guides what actions need to happen, when and by whom.

The Victorian Government takes a risk management approach to managing invasive animals, which is aligned with national *Guidelines for the Import, Movement and Keeping of Non-indigenous Vertebrates in Australia* (Vertebrate Pests Committee 2014) and is focused on the primary objective of protecting Victoria from their threats and impacts. The management of existing and potential invasive species in Victoria is guided by a whole of government *Invasive Plants and Animals Policy Framework* (IPAPF) (Victorian Government 2010).

The IPAPF emphasises a risk-management approach to government involvement and investment to ensure that Victoria is well-positioned to meet future biosecurity challenges. At the centre of the IPAPF is the 'invasion curve' (Victorian Government 2010), which outlines the different stages of invasive species management: Prevention, Eradication, Containment and Resource Protection and Long-term Management. The 'invasion curve' from IPAPF has been widely adapted and used for biosecurity applications nationally and internationally, including in the development and improvement of national biosecurity policy (Australian Weeds Committee and the Vertebrate Pests Committee 2012; Craik *et al.* 2017). The ability to prevent and/or rapidly respond to an alien species is at the heart of the non-indigenous bird reforms in Victoria (see case study).

Government has a variety of tools that influence or control the way people behave. The main legislation covering invasive animal management in Victoria is the *Catchment and Land Protection Act 1994* (CaLP Act). It provides the power to declare animal species as belonging to one of four pest animal categories (prohibited, regulated, controlled or established) to protect primary production, Crown land, the environment and community health from the threats posed by the species. Declaration enables the regulation of control, importation into the state, keeping, movement, trade and release of pest animals in Victoria. The CaLP Act is formally

Case study: Rose-ringed Parakeet in Victoria

Without appropriate intervention, it is only a matter of time before the Rose-ringed Parakeet *Psittacula krameri* (Fig. 1), becomes established in Australia (Vall-Ilosera *et al.* 2017). It has established feral populations in 35 countries on five continents (Butler 2003; Menchetti *et al.* 2016). What makes this species a remarkable invader is that it is not confined to its tropical origins and has successfully established in cool climates in Europe (e.g. United Kingdom and the Netherlands; Butler 2003).

The Rose-ringed Parakeet is a charismatic, gregarious, tropical Afro-Asian bird that is held extensively in private collections in Australia. If established in the wild, it could pose significant risks to the economy, environment and social amenity of Victoria (Vall-Ilosera *et al.* 2017). The Rose-ringed Parakeet could threaten biodiversity through competition for nest hollows and food resources used by native parrots such as rosellas and other hollow-nesting birds.

Vall-Ilosera *et al.* (2017) observed 861 parakeet incursion events in Australia, involving at least 1151 individuals for the period 1999–2013. Most of these observations were from 2011–2013 because of the online role of the Australian Rescue and Rehoming Resource (Vall-Ilosera *et al.* 2017). The authors observed that most of these reports occurred within the community keeping the birds and were not reported to the appropriate government biosecurity authorities.

These data also coincided with an increase in the availability and reach of social media, making reporting easier. What can be concluded from their study is that the propagule pressure is high. In Western Australia, some escaped birds are thought to have survived in the wild for at least four years, and two groups had reportedly bred (Department of Primary Industries and Regional Development, unpublished data). Several captured birds had leg bands that identified them as originating in captivity.

Recognising the threat posed by the Rose-ringed Parakeet, Vall-Ilosera *et al.* (2017) advocated for an integrated approach that includes awareness programs involving local communities, the interrogation of novel data sources and traditional biosecurity surveillance for detecting new incursions, and rapid response once breeding in the wild is detected. The non-indigenous policy, and associated declarations, now provides the Victorian Government with the necessary powers to undertake all reasonable steps to control a population of Rose-ringed Parakeet on any land in the State of Victoria, with the goal of preventing this species from establishing.



Fig. 1. The Rose-ringed Parakeet is held extensively in private collections in Australia, but if established in the wild, it could pose significant risks to the economy, environment and social amenity. Photo Tim Blackburn.

administered jointly and severally by the Minister for Energy, Environment and Climate Change and the Minister for Water. The Minister for Agriculture is responsible for invasive animal policy and direction setting, while her department (Department of Jobs, Precincts and Regions) has been delegated the responsibility for taking all reasonable steps to control animals declared in the prohibited, controlled or regulated categories on any land in the state. Each public and private land manager is responsible for taking all reasonable steps to

prevent the spread of, and, as far as possible eradicate, established pest animals on his or her land. The Department of Jobs, Precincts and Regions has Authorised Officers who are responsible for enforcing landholder compliance with the pest animal (and noxious weed) provisions of the CaLP Act.

In 2014, no species of birds had been declared as pest animals under the CaLP Act. A clear and robust direction was set for the management of non-indigenous birds to redress this significant gap in invasive animal policy.

The policy

The policy approach, and associated regulations, needed to be mindful of several factors to assist the Victorian Government to better manage the risks and impacts of non-indigenous birds in Victoria. The outcomes-based policy needed to be simple, not impose a regulatory burden and be consistent with the IPAPF. The policy had five key elements:

1. Non-indigenous bird species not known to be in Australia

Non-indigenous bird species are declared as prohibited pest animals under the CaLP Act if the species meets the following criteria:

- The species is not known to be present in Australia;
- the national Environment and Invasives Committee (EIC), or its preceding committees (i.e. the Vertebrate Pests Committee followed by the Invasive Plants and Animals Committee) has categorised it as being of extreme or serious threat to Australia as per the Bomford (2008) risk assessment model.

Should the Australian Government amend the Live Import List (*Environment Protection and Biodiversity Conservation Act 1999*) to allow import of these species into Australia, they would be prohibited from being brought into, kept or sold in Victoria. The Victorian Government may consider applications to keep prohibited pest animals for medical, veterinary or biological research purposes where there is no alternative, non-declared species that could be used. This approach will minimise the chance of such species becoming established in Australia and particularly Victoria. Should birds of these species be detected in Victoria, Agriculture Victoria will take all reasonable steps to eradicate the species from the state.

2. Non-indigenous bird species restricted to approved Victorian zoological facilities

Non-indigenous bird species will be declared as controlled pest animals under the CaLP Act, and their keeping limited to approved zoological facilities in Victoria, if they meet the following criteria:

- The species is present in Australia but is found only in recognised zoological facilities such as zoological parks declared under the *Zoological Parks and Gardens Act 1995* (e.g.

Melbourne Zoo, Werribee Open Range Zoo and Healesville Zoo) or non-statutory zoos and wildlife parks (e.g. Halls Gap Zoo and Ballarat Wildlife Park);

- it has been categorised and endorsed nationally as being of extreme or serious threat to Australia; and
- it has complex husbandry requirements.

Birds that are already restricted to approved zoological facilities in Victoria will be considered for declaration as a controlled pest animal. This approach prevents such species being kept widely in Victoria and minimises the chance of them escaping or being deliberately released. Should birds of these species be detected in Victoria, Agriculture Victoria will take all reasonable steps to eradicate the species from the state.

3. Non-indigenous bird species that are known to be in Victoria in private collections

The government does not consider it reasonable to increase the regulatory burden for existing bird keepers in Victoria. Permits to keep species already known to be in private collections would not be required. Bird species known to be in private collections in Victoria will be allocated as described in 3a and 3b based on their threat to Victoria.

3a: Non-indigenous bird species that are kept in Victoria and are determined to pose a high threat of establishment in the wild in Victoria

The feral or wild populations of non-indigenous bird species will be declared as regulated pest animals under the CaLP Act if the species meets the following criteria:

- The species is not widely established in the wild in Victoria;
- EIC has categorised it as an extreme or serious threat to Australia, and/or the species is documented to be an escapee in other Australian jurisdictions; and
- there is a likelihood that government intervention will lead to eradication or containment of feral or wild populations of the species.

Captive populations of these species will not be declared and will not be subject to regulation. However, birds of these species that are observed in the wild (i.e. outside of a secure enclosure) will be deemed by the government to be feral or wild and, in the broader public interest, will be subject to control measures. These

incursions usually involve only a few individual birds (escapes from private collections). The Victorian Government will manage incursions in accordance with the IPAPF and its duty to take all reasonable steps to control the animal on any land in the state, with a priority given to those species that have a reasonable chance of successful eradication with the resources available.

3b: Non-indigenous bird species determined not to pose a high threat of establishment in the wild in Victoria

The government will have no role in the active management of birds that are unlikely to survive in the wild, but may provide advice for the management or recapture of these animals when detected in the wild.

4. Non-indigenous quail, pheasants and partridges, classified as game species

Game species are managed under the *Wild-life (Game) Regulations 2012*. The Victorian Government did not consider it reasonable to change the current management of non-indigenous gamebirds, i.e. quail, pheasants and partridges.

5. Non-indigenous bird species that are widely established in Victoria

The Victorian Government did not consider it reasonable to impose the lawful responsibility to control widely established bird species upon landowners (including those in urban areas). The Victorian Government will support relevant research and the provision of advice, for the management of these species in the wild, where appropriate.

Stakeholder feedback

As with any appropriate policy development, we sought feedback from the community as the policy was developed, and again when species were being identified for declaration. A discussion paper was released in 2013 on the draft policy, supported by a series of workshops and a webinar. Consultation commenced in May 2013 and ran for six weeks. A total of 72 individuals from 69 organisations were invited to participate in the consultation, with 25 individuals (23 organisations) participating in the briefing sessions/webinar. A total of 17 submissions

were received during this process.

During this initial engagement, all stakeholders were positive about the engagement and consultation process. Stakeholders appreciated that they had been engaged early in the process and contacted individually. Submissions were generally supportive and constructive, although there was a diversity of views. All stakeholders recognised the need to have policy to manage threats from non-indigenous birds. Some stakeholders believed that the proposed approach would not be strong enough to reduce risks from non-indigenous birds, yet others believed that some elements may be too prescriptive. Even so, there was general agreement on the importance of preventing high-threat-potential species from entering the state and that the Victorian Government has a key role in managing high-threat, non-established, species in the wild.

In 2014, we also provided stakeholders with the opportunity to comment on the list of bird species proposed to be declared under the CaLP Act. The proposed list for declaration assigned species to the CaLP Act classifications of prohibited pest animal (n=1), controlled pest animal (n=3) and regulated pest animal (n=37).

Fifty-one submissions were received concerning the species proposed to be declared. Of the respondents, 46 were from the avicultural community and were generally unsupportive of the proposed list of species to be declared. Their responses could be categorised as:

- birds from a captive origin would not survive in the wild and are a low risk (n=24);
- the focus of the list should be on established species such as the Common Myna (n=7);
- concern about particular individual species on the list (n=5);
- concern over the validity of the risk assessment process (n=4); and
- concern that declaration would lead to licensing or confiscation of birds (n=4).

Of the five favourable responses, two suggested a more prescriptive risk-based approach was needed and another suggested that the policy be broadened to include animal welfare.

Species declared

The response from the stakeholders was considered and the list was revised, with

Governor-in-Council, on recommendation of the Minister (the then Minister for Environment and Climate Change), declaring 19 species of birds in line with the policy as pest animals under the CaLP Act.

One species, the House Crow *Corvus splendens*, was declared as a prohibited pest animal in recognition of the importance of keeping this agricultural and environmental pest out of Australia.

Two species, the Greater Flamingo *Phoenicopterus roseus* and the Greater Rhea *Rhea Americana*, were declared as controlled pest animals. These species can be kept in Victoria, but are permitted only in high security collections such as zoological parks.

The feral and wild populations of sixteen species were declared as regulated pest animals: Egyptian Goose *Alopochen aegyptiacus*; Canada Goose *Branta canadensis*; African Collared-dove *Streptopelia roseogrisea*; Northern Bobwhite *Colinus virginianus*; Red Avadavat *Amandava amandava*; Common Waxbill *Estrilda astrild*; Tricoloured Munia *Lonchura malacca*; Java Sparrow *Lonchura oryzivora*; Red-cheeked Cordonbleu *Uraeginthus bengalus*; Red-whiskered Bulbul *Pycnonotus jocosus*; Peach-faced Lovebird *Agapornis roseicollis*; Monk Parakeet *Myiopsitta monachus*; Nanday Parakeet *Nandayus nenday*; Alexandrine Parakeet *Psittacula eupatria*; Rose-ringed Parakeet (Indian Ringneck Parakeet) *Psittacula krameri*; and Ostrich *Struthio camelus*. The captive populations of these species are not declared, meaning these species can continue to be kept and traded in Victoria.

Discussion

The policy development process raised some important issues, particularly about risk management of birds and their potential impacts. Some stakeholders were adamant that there is no risk of birds escaping from captivity and becoming established in the wild. One stakeholder suggested 'just because a thing could happen doesn't mean it ever will'. Clearly, this is not the case. Rainbow Lorikeets *Trichoglossus moluccanus*, escaped from captivity or were deliberately released in Perth, Western Australia, in 1968 and are now firmly established (Chapman 2005). Likewise, the Case Study of

the Rose-ringed Parakeet demonstrates that escapees from captivity are common and widespread, and reported to peers rather than government. Moreover, the Rose-ringed Parakeet has the potential to impact negatively on native species (see Case Study), so it is important that the risks are mitigated, which the policy enables.

Communication and engagement were an important part of developing policy and implementing the changes. Stakeholder engagement was central to this process, but this does not mean that we changed views or concerns. For example, concern that the policy did not focus on the established species reflects the challenges of communicating what this would mean under the CaLP Act (i.e. it would require the landholder to undertake management of the declared species). Likewise, the subtlety of declaring feral or wild populations of some species as Regulated pest animals caused unnecessary concerns for keepers of those species. More work is required in the human dimensions of regulatory practices and wildlife management where there are conflicting views. From our perspective, the policy and associated declarations of non-indigenous birds can be considered a success in terms of balancing the views of the diverse stakeholders and achieving the necessary biosecurity outcomes we set out to achieve. The policy enables prevention and early intervention, which previously was not possible. This means that the Victorian Government now has a mechanism to manage potentially harmful species, such as the House Crow. From an aviculturist's perspective, the policy has not imposed a regulatory burden or restricted the keeping of non-indigenous birds. For the broader community, the policy and declarations have not imposed costs on landowners to control established pest species, such as the Common Myna. As a measure of success, there was minimal adverse correspondence to the Victorian Government upon implementation of the policy and regulations.

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Community-led approaches can lead to better outcomes for management of European Rabbits *Oryctolagus cuniculus* and other invasive species

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Abstract

European Rabbits *Oryctolagus cuniculus* are incredibly devastating pest animals in Australia and New Zealand. For over 150 years, different approaches to managing impacts of Rabbits have been considered with varying levels of success and longevity. The focus of Rabbit management has traditionally been directed towards mitigating the impacts on agricultural production, where the economic costs are significant. Negative environmental, social, and cultural impacts are often overlooked. We briefly report on the history and problem of Rabbits and their management, leading to the focus of the paper—a community-led approach to managing Rabbits via the Victorian Rabbit Action Network (VRAN). This approach considers that the key to successful strategies for management of European Rabbits is to understand the stakeholders. More specifically, it is about understanding the individuals and organisations involved in Rabbit management and having a deep appreciation of their interests, needs, knowledge and experience, and of the political and cultural dimensions. This approach to Rabbit management brings together different types of knowledge, experiences and perspectives to address a common problem. It fosters creativity and innovation, is flexible, nimble and agile, and coordinates limited resources effectively. This inclusive approach, which has broad reach across rural and regional communities of Victoria, has resulted in a positive change in mind-set and practice. VRAN may be considered a blueprint of what can be achieved for all system participants through shared responsibility in addressing a significant biosecurity issue. It offers a mechanism whereby significant issues facing our society, such as the relationship between biodiversity, climate change, loss of habitat and invasive alien species, can be discussed and acted upon. (*The Victorian Naturalist* 137 (6), 2020, 210–219)

Keywords: community, engagement, invasive species, Rabbits, wicked problems

Introduction

The European Rabbit *Oryctolagus cuniculus* in Australia and New Zealand is an incredibly devastating mammalian invader. Only the House Mouse *Mus musculus* exceeds the abundance of Rabbits in Australia as an invasive alien mammal (Williams *et al.* 1995). Indeed, Rabbits are considered the most widespread and abundant wild herbivore in southern Australia (Mutze 2016) and have a major presence and impact in every state. The sheer number of Rabbits in Australia, their broad distribution across much of the continent, combined with their herbivory and semi-fossorial habits means that this species poses a significant threat to biodiversity

through land degradation and competition with native herbivores (Reddix *et al.* 2007; Commonwealth of Australia 2016a).

Over the past 150 years, management of Rabbits in Australia has primarily focused on mitigating their economic impacts (i.e. negative impacts on agriculture). Soon after Rabbits were successfully introduced to Victoria in 1859, authorities across Australia recognised that they seriously impacted Australia's primary production. The speed, scale, and extent of the reaction by authorities to manage Rabbits was unheralded. The first legislative tool, *The Rabbit Destruction Act 1871*, was

introduced on 21 December 1871 to 'provide for the destruction of Rabbits in Tasmania' (Stoddart and Parer 1988), followed by similar legislation in other jurisdictions in subsequent years. Recognising that continent-wide spread was inevitable, twenty-six years later the Western Australian Government took action to prevent Rabbits expanding into that state. The world's longest fence was started in 1901 and completed in 1907 to prevent the westward movement of Rabbits into the pastoral areas of Western Australia (Williams *et al.* 1995). This was a significant capital investment in prevention, which turned out to be unsuccessful. Australia's Rabbit-proof fences and their remnants today are testimony to the risks posed by, and consequences of, this species' presence.

According to contemporary estimates, Rabbits cost rural economies across Australia more than \$200 million per annum (Gong *et al.* 2009). This impact cost estimate is typically framed in terms of lost agricultural production, and, to a lesser extent, cost of control (Reddiex *et al.* 2007). The economic analyses of McLeod (2004) and Gong *et al.* (2009) have been used to quantify the monetary impact of Rabbits. These provide supporting evidence for additional research and development, particularly for biological control methods in Australia, such as boosting Rabbit biocontrol via an additional variant ('K5') of Rabbit Haemorrhagic Disease Virus (RHDV1) (Strive and Cox 2019).

With the focus on managing the economic impacts of Rabbits, a variety of tools and techniques have been developed, applied, and adapted. These include the introduction of legislation over time (e.g. *The Rabbit Destruction Act 1871* in Tasmania, the *Rabbit Destruction Act 1875* in South Australia, the *Rabbit Suppression Act 1880* in Victoria and the *Rabbit Act 1902* in Western Australia), a suite of extension products and tools to inform communities how to manage Rabbits (Williams *et al.* 1995), chemical control tools (e.g. 1080, pindone, fumigation), physical control tools (e.g. warren ripping, ground shooting), and biological control (e.g. RHDV1 and myxomatosis) (Williams *et al.* 1995; Brown 2012). These tools have been very effective at reducing Rabbit numbers and negative impacts, and produce associated economic and environmental benefits (Cooke *et al.*

2013). However, the inexorable reproductive biology of Rabbits has made such effects ephemeral over time. Rabbit populations are not static; they move, and are thus a common property problem, a 'wicked' one at that. Hence, a collaborative, coordinated landscape-scale approach to management is necessary. The key to success in our experience is community participation, engagement and empowerment (Adams *et al.* 2019; Reid *et al.* 2019), so that control programs are locally relevant, 'owned' by the community, and more resilient to shifting government priorities and staff turnover. Empowering communities to take ownership of Rabbit management is the main focus of this paper.

Rabbits are a significant environmental problem

Rabbits are one of the most widely distributed pest animals in Victoria, occurring in a wide range of habitats. They are found from sub-alpine regions through to the arid landscape of the Mallee, throughout the central ranges and grasslands through to the southern and eastern coastal plains. Soil is a major indicator of susceptibility to Rabbit infestations, with the species preferring deep and sandy soils. Non-arable rough country, which includes creeks and river banks, erosion gullies, rocky outcrops and forest grassland interfaces, is particularly susceptible to high Rabbit densities (Myers *et al.* 1994). The Rabbit population across Victoria fluctuates significantly as a result of factors such as breeding events, the impacts of bio-control agents or extreme climatic events (e.g. droughts), and availability of feed (Commonwealth of Australia 2016a).

Rabbits have a disastrous impact on Australian flora and fauna, competing with wildlife for food and shelter, damaging native plants through grazing, and preventing the regeneration of seedlings. Figures 1–5 provide examples of the severity of damage Rabbits can cause. Rabbits impact over 300 species listed as threatened under the *Environment Protection and Biodiversity Conservation (EPBC) Act*, including 44 fauna species and 260 plant species (Commonwealth of Australia 2016b).

According to Williams *et al.* (1995), decades of intensive grazing pressure by Rabbits (alongside other introduced animals) has likely



Fig. 1. Evidence of Rabbit grazing causing ringbarking of tree.



Fig. 2. Paddock overgrazed by Rabbits and grazing-height of tree foliage demonstrated.



Fig. 3. Disturbed soils with weeds result from Rabbits selectively grazing and removing palatable pasture and tree species.



Fig. 4. Impact of Rabbit grazing on seedling.



Fig. 5. Rabbit grazing on new planting.

permanently altered Australian landscapes, with many grass and herb species having disappeared prior to the introduction of Rabbit biological control. The same authors reported that the extent of damage to rangeland shrubs and trees is often masked by their long life span and episodic regeneration; however, the replacement rate of ecologically sensitive rangeland vegetation communities is often insufficient to prevent their disappearance in the long-term, even when Rabbits are present in low numbers. Bird *et al.* (2012) reported damaging impacts on she-oak regeneration at much lower densities than previously reported and suggested that the low-density threat is likely to be mirrored in many other tree and shrub species. The review conducted by Williams *et al.* (1995) suggested that for many of the more palatable native species—for example, some species of *Acacia*, *Stipa* and *Danthonia*—there may be no safe Rabbit density, with regeneration observed only when Rabbits are completely excluded.

Beyond their direct impacts on seedling survival, Rabbits also significantly damage native animal populations through competition for food resources, habitat damage, and indirect impacts on predator populations. Cooke and Mutze (2018) reported on the similarities between the quality of foods selected by Rabbits and those needed by young kangaroos, and observed increases in kangaroo numbers following removal of Rabbits. Bird *et al.* (2012) showed increases in both kangaroo and wombat numbers following Rabbit control, but that the higher density of both genera had no measurable effect on native seedling survival rate. Pedler *et al.* (2016) reported that the release of RHDV1 has been the single most important factor in dramatic increases in the numbers of several small threatened mammals in arid inland Australia, due to decreased competition for food resources and declines in Rabbit-dependent predators. A lag-time between a decline in Rabbit numbers and the corresponding drop in predator numbers can result in intense predatory pressure on native mammal species for a short period after the Rabbit crash. If the Rabbit reduction is sustained through ongoing management, this pressure occurs once; if Rabbit numbers 'yo-yo' due to improved seasonal conditions or lack of ongoing management, the

predatory pressure happens again and again, causing severe and lasting damage to native mammal populations (Williams *et al.* 1995).

Reframing the Rabbit problem

There is no easy solution to the Rabbit problem—Rabbit management is a classic 'wicked' problem (Rittel and Weber 1973) with its tangle of economic, ecological, sociological, and political influences. Management of Rabbits in Victoria is complex, in part because there are so many organisations involved, each with differing agricultural and environmental objectives. This situation has resulted in the wide distribution of investment across organisations, each with their own institutional arrangements and politics. The structure of our public institutions often does not allow integrated programs across the triple bottom line (agricultural/economic, social and environmental outcomes), meaning environmental and social drivers are often lower in priority than agricultural ones.

A range of effective, scientifically proven Rabbit control methods are available (Williams *et al.* 1995). But across any given landscape, the acceptable level of Rabbit density differs depending on the outcome sought: in cropping areas, a few Rabbits might be acceptable from a production (agricultural) perspective, whereas complete eradication of Rabbits is fundamental to the environmental drivers on Macquarie Island (Springer 2016). Similarly, densities of less than one Rabbit per hectare can continue to suppress native seedling survival in sensitive conservation areas (Bird *et al.* 2012) but may be enough to meet landholders' regulatory requirements under the *Catchment and Land Protection Act 1994*. Moreover, these regulatory requirements might protect agricultural values, but they may fall short of achieving environmental protection. Managing Rabbits for environmental outcomes typically requires more intense management inputs.

Across Australia there has been an institutional shift towards a model of 'shared responsibility' for biosecurity (Council of Australian Governments 2019), including for management of invasive species such as Rabbits. This shift recognises that governments cannot act alone to deliver or enforce the sustained and coordinated efforts that are needed for effective

control (Martin *et al.* 2016), and that greater community involvement and collaboration is needed. But despite the strength of the control methods and technologies available to manage Rabbits, there is no comparably robust framework for managing the human dimensions of Rabbit control (Martin *et al.*, 2016)—that is, the human behavioural and social capacity elements that determine the likelihood of individuals and communities to initiating and maintaining an effective Rabbit control program.

In addition to a shift towards 'shared responsibility' (Council of Australian Governments 2019) there are some systemic socio-economic and cultural forces afoot in rural and regional Australia that play significantly into the dynamic of managing invasive alien pests. Some of the key changes that impact invasive species management include diminishing resources (declining public investment), changes in government priorities, less capability (fewer people) and capacity (declining knowledge base), and a growing disconnect between rural and urban communities. These changes, and shifting cultural norms associated with them, can be seen in communities where 'tree changers' have limited knowledge or interest in land management obligations (Ragusa 2010), or communities where there is a skewed age demographic (O'Callaghan and Warburton 2017) which is then reflected in the activities of community Landcare groups.

A community-centred approach to Rabbit management

Given the Rabbit's prolific breeding and mobility, coordinated and cooperative approaches to managing Rabbits are imperative. Yet across the socio-political system, there are multiple stakeholders across both public and private institutions. These stakeholders have varying interests and capacities, and differing incentives and constraints that affect their participation in addressing Rabbit management problems. Because of these complex arrangements and barriers, aspirational approaches to Rabbit management, such as tenure blind (e.g. Braysher *et al.* 2012), could be considered unachievable. Instead, a facilitating organisation mechanism is required to support a co-operative and collaborative approach involving both public and

private land managers. The status quo approach to Rabbit management is a regulatory paradigm of command and control, i.e. enforcement. A new paradigm is needed—one which is disruptive to the status quo approach and reflective of contemporary community expectations and available resources.

In this context, our first step was explicitly to embrace the complexity of Rabbit management in Victoria. We deliberately identified Rabbit management as a 'wicked' problem (Adams *et al.* 2019), with no single solution; responsibility for management resting with multiple actors; and no precise definition of the issue (Head and Alford 2008). This framing recognised the inherent contestability and political nature of the 'Rabbit problem' and the need for sustained behavioural change on the part of multiple parties (Australian Public Service Commission 2012) as central to effective management and control. Rather than focusing on either agricultural or environmental outcomes, our approach was to deepen the participation of all stakeholders, particularly community, to develop a community-led approach. In concert with community partners and private landholders, we sought to co-create a model of cooperative governance, ensuring that people most affected by Rabbits are central in the process of defining the problems, co-creating strategies to address these problems, and sharing in investment decisions. Through this collaborative effort, the Victorian Rabbit Action Network (VRAN) was co-created as an enabling structure for Rabbit management in Victoria. Under this model, multiple initiatives were developed to build capacity and capability of the people who manage Rabbits through training, community engagement and promotion of best practice control techniques.

Victorian Rabbit Action Network (VRAN)

VRAN was established as a vehicle to reframe the collective thinking about the Rabbit problem and how it can be managed. It is a cooperative institution between citizens and government, conceived and built on the basis that Rabbit management in Victoria can be improved by bringing everyone together. Such improvement stems from the differing expertise, experiences and insights that people who do not normally

work together bring to the discussion and collaboration. This ‘new’ or non-redundant information is a source of learning, creativity, and innovation for individuals and across the group. As a facilitating entity, VRAN provides the strategic mechanism to bring community voices and experiences into the design and development of programs. This recognises that community-led approaches are as much about social empowerment as they are about solving technical problems. Community-led approaches can improve the resilience and effectiveness of management programs by building social capacity, bringing local knowledge and experience to bear, changing institutional structures and processes, and shifting to shared decision-making. VRAN provides a mechanism to enable more integrated, inclusive, and constructive politics among those involved (government, non-government and private organisations, and individuals). The politics and relationships formed through the facilitating processes of VRAN serve to sharpen attention and thinking on the desired outcomes and how to achieve them. Inevitably, the outcomes address managing the impacts of Rabbits for economic, environmental, and social value rather than just one of these elements.

A key strength of the VRAN approach is its focus on enabling outcomes by responding to needs identified by the participants themselves, largely through communication, training, education, coordination, networking and working to a common goal. That is, learning how to apply a control tool in the right circumstance is a core skill that is required regardless of whether the objective is to protect biodiversity or agricultural land. Collectively understanding the nuances associated with the application of management tools for biodiversity (e.g. not

being able to warren rip an ecologically sensitive area) or cultural protection (e.g. protecting Aboriginal burial sites) enables communities and governments to reach a common understanding of what needs to happen and how to achieve it. The approach has brought together stakeholders that once worked independently at best or against each other at worst.

Emerging impacts of VRAN

As with any disruptive initiative, it is important to measure the change that can directly be attributed to VRAN. In an independent impact analysis (ACIL Allen 2017), the two most significant changes attributable to VRAN were an increase in knowledge and awareness of the need for coordinated Rabbit management and the development of supportive networks to deliver effective Rabbit management. VRAN has increased knowledge and collaboration, built confidence in best-practice Rabbit control, improved relationships and networks across different parts of the Rabbit management system, and changed mindsets about how institutions and community groups can work together (Table 1).

The cooperative governance model ensures that people most affected by the Rabbit problem are central in the process of defining the problems, co-creating strategies to address these problems, and sharing in investment decisions. Through VRAN, there has been a significant shift in governance for Agriculture Victoria, one requiring government biosecurity directors and program officers to engage democratically with citizens in sharing decision-making responsibility and power for Rabbit management—in effect, to work as democratic professionals (Dzur 2018). Now, instead of Agriculture Victoria delivering its programs to

Table 1. Impact of VRAN in first three years of establishment. Source: ACIL Allen (2017).

Key observations

Reach	Engaged more than 5200 people directly and indirectly 34% of these people were engaged with Landcare
Change in mind-set	55% of people changed their views on collaboration 59% of people have been inspired to increase rabbit management activities 55% of people have increased confidence to manage Rabbits
Practice-change	Over 80% of people have made changes to how they consider integrated approaches to rabbit management

the community, it is working with the community to design programs, so that they are locally supported, relevant and more effective.

Case Study—Neds Corner

Neds Corner Station is the largest freehold property in Victoria. It is an example of the ecologically significant Victorian mallee habitat that is extremely sensitive to intensive grazing from livestock, native herbivores and Rabbits (Sandell 2002).

In 2002, Neds Corner was purchased by the Trust for Nature in recognition of its biodiversity, geological and Aboriginal values. Trust for Nature set about removing livestock and undertaking a program of extensive Rabbit and European Red Fox control, as well as large-scale revegetation activities. Now, what was once described as 'bare sand hills, grazed by thousands of Rabbits which weren't allowing any native regeneration' (P Barnes, manager Neds Corner Station, Trust for Nature) is vegetated with salt-bush and bluebush vegetation. At the peak of the Rabbit problem, spotlight counts were averaging 30–35 Rabbits per kilometre, and after a decade of work, the average is now 0.4 Rabbits per km.

VRAN has leveraged and added to the success of Rabbit management at Neds Corner, with flow-on benefits for the broader Mallee ecosystem. The manager of Neds Corner was one of the first graduates of a VRAN training program on Rabbit management (called a bootcamp). The bootcamp laid the groundwork for developing an ongoing community of practice called the Leaps and Bounds learning network. This network meets regularly to share information, experience, and expertise on Rabbit management. Through the bootcamp and the Leaps and Bounds learning network, the manager of Neds Corner felt his approach changed from killing Rabbits to a more strategic approach of managing their impacts. The success of Neds Corner as an exemplar of Rabbit management and the advocacy of the VRAN approach has had flow-on benefits to the broader community (ACIL Allen 2017). Sharing of experiences on the complexity and challenges of managing Rabbits on a large scale, combined with advocacy of continual learning and reflection, empowers other communities that are dealing with similar issues (Muth *et al.* 2019).

Discussion

It is important to frame our work in terms of contemporary biosecurity policy in Australia. The first principle of the Intergovernmental Agreement on Biosecurity (IGAB) is that 'Biosecurity is a shared responsibility between all system participants' (Council of Australian Governments 2019). Although 'shared responsibility' is central to our approach to Rabbit management, there is a tendency for this term to become overused and therefore unhelpful. Moreover, there is often limited attention paid to the second part of the principle being 'all system participants'. The focus of our work has been on both the 'shared responsibility' and the 'system participants', which we consider to be essential for complex biosecurity issues such as Rabbit management. As Australia's key national biosecurity policy, IGAB is not particularly informative about how its principles can be implemented by the broader community or system participants. Therefore, we believe that VRAN could be considered a blueprint of what can be achieved for shared responsibility of a significant biosecurity issue in Australia.

VRAN is an important enabler of 'shared responsibility' and is a facilitating organisation for the 'system participants'. It brings together people with different ideas, experience, and expertise. VRAN has made it possible to learn creative and innovative activities in a safe, non-judgemental environment. Through VRAN's network-building function, the many stakeholders involved in Rabbit management in Victoria now have a more accurate working knowledge of what is happening in practice and how community action on Rabbits can be made more effective. As a result of VRAN, a wider range of knowledge, expertise and experience can now be harnessed for determining interventions that improve the quality of Rabbit management in ways that are locally relevant. Environmental objectives are treated in the same manner as agriculture protection. Mechanisms enabled by VRAN, such as training, education, learning networks, community engagement and so on, have broad reach in terms of numbers of people and geographical regions (Table 1).

Another important principle of IGAB is that 'System participants are involved in planning and decision-making according to their roles, responsibilities and contributions' (Council of Australian Governments 2019). Shared responsibility and decision-making requires sharing of power. VRAN has shifted power relationships and dynamics for all involved, and such shifts can be empowering for some interests and threatening for others. It is unavoidable that the matters that VRAN addresses are politicised. Diversity of opinion is critical for a democratic process. We have found that shifting the power dynamic is a key part of the disruptive nature of VRAN. It is also a key reason for VRAN's success and more reflective of true system participation.

At the heart of VRAN are relationships. The relationships stemming from VRAN have not only been instrumental to VRAN's success but have also prepared the people, organisations, and communities involved to deal more quickly and more effectively (because they know each other) with other unrelated issues, both known and unimaginable, that will confront them in the future. This may be the real legacy of VRAN: preparation for handling the radical uncertainty (Kay and King 2020) that is the existential reality of life and living in our world. VRAN offers the vehicle where the relationship between biodiversity, climate change, loss of habitat and invasive alien species (Pereira *et al.* 2012) can be discussed and acted upon.

We believe the VRAN approach can be applied to other complex issues concerning invasive species: many stakeholders; diverse and contested views; shortage of funds; and, as noted, complicated lack of, and/or failure of regulatory or policy tools. This approach is important where there is discrepancy in the value of investment across the triple bottom line and/or real or perceived lack of leadership from government (Martin *et al.* 2016). For Victoria, the management of deer (Davis *et al.* 2016) would be an ideal application.

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Eradication of Feral Pigs on Quail Island to protect and restore ecological values

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Abstract

Feral Pigs (*Sus scrofa*) were illegally released onto Quail Island in 2008 and subsequently caused damage to soil and vegetation on the island. Damage observed across the 700 ha island included pugging, creation of wallows, competition for water, creation of trails, promotion of weed spread and predation of wildlife. Control activity undertaken periodically since 2013 was scaled up in 2019 to include aerial and ground shooting. No Feral Pigs were detected on the island in the twelve months that followed ground shooting, suggesting that the population had been eradicated. This paper reports on the procedures required to eradicate a small Feral Pig population on an ecologically sensitive island. (*The Victorian Naturalist* 137(6), 2020, 219–227)

Keywords: island restoration, Ramsar, invasive species, camera detections, aerial and ground shooting, poison baiting

Introduction

Quail Island (38.234003S, 145.291267E), located within the North Western Port Nature Conservation Reserve, is a Ramsar Convention-listed wetland of international importance due to its ecological, botanical, zoological and

hydrological importance. Significant vegetation communities include herb-rich woodlands, swampy scrubs and woodlands, heathlands, mangrove, samphire shrubland and saltmarsh communities. Salt Lawrenia *Lawrenia spicata*

and Yellow Sea Lavender *Limonium australe* are present within the saltmarsh communities and listed as rare in Victoria. A fauna survey in 2008 indicated a significant population of Southern Brown Bandicoot *Isodon obesulus* was present (Legg 2010; MacLagan 2016). Migratory waders such as Caspian Tern *Hydroprogne caspia* and Ruddy Turnstone *Arenaria interpres* frequent the mangroves and mudflats fringing the island.

The island was grazed from the 1860s into the 1900s, supported by two excavated freshwater dams that have since become ephemeral freshwater water points (Patterson 2015). The northern-most water point forms the most reliable freshwater point on the island. Koalas *Phascolarctos cinereus* were introduced in 1933 and rapidly overbrowsed the island and remain common today (Menkhorst 2008). Red Fox *Vulpes vulpes* and European Rabbit *Oryctolagus cuniculus* breed successfully on the island.

A pregnant Feral Pig *Sus scrofa* was released onto Quail Island in 2008 by unknown person(s), presumably for hunting purposes (S Coutts, pers. comm. 1 July 2020). Further releases in 2010 and 2012 were recorded, with follow-up baiting and shooting used by contractors and Parks Victoria to remove at least 17 Feral Pigs between 2013 and 2015 (M Legg, pers. comm. 16 July 2020). Landowners of properties adjoining the coastline immediately to the north of Quail Island reported Feral Pigs accessing their properties, particularly at dams during summer. Trail cameras subsequently confirmed that the population was breeding, with three piglets being observed in 2016. Based on camera detections, the population was estimated to include 12–16 individuals in 2018.

Damage attributable to Feral Pigs on Quail Island was readily identifiable on saltmarsh as well as throughout the interior of the island. Wallows were created in areas in moist soils (freshwater) as well as throughout areas subject to tidal inundation (Fig. 1). The sandy banks surrounding the water point were extensively disturbed. This area was heavily pugged and reduced to dried mud in summer, with the pigs maintaining a deepening pit to access water. Other sandy soils were commonly turned over to a depth of 300 mm. Through dense vegetation across the island, Feral Pigs created



Fig. 1. Examples of pugging damage and wallow caused by Feral Pigs on Quail Island. Photos A Bolden.

trails that were subsequently exploited by other species, including foxes. Weed species such as Ragwort *Jacobaea vulgaris*, syn. *Senecio jacobaea* have established in areas subjected to soil disturbance by Feral Pigs.

Given the small population size, eradication of the Feral Pig population was an achievable objective and would remove a key threat to the environmental values on the island. This paper reports on the progression of techniques used to remove the Feral Pigs on Quail Island.

Methods

In November 2015, a monitoring operation was conducted to estimate the size of the Feral Pig population and inform future management decisions. Six trail cameras were set around the water point as well as in other areas of high visitation, given that pigs must access fresh water daily (Choquenot *et al.* 1996). Locations of Feral Pig activity, including rooting, track creation and scats, were recorded.

In January 2016, Quail and Chinaman Islands were reclassified from State Faunal Reserves into the North Western Port Nature Conservation Reserve, under the *Crown Land (Reserves) Act 1978*. This reclassification provided regulations to help manage illegal activities on the islands. The change in reserve status was advertised using regulatory signage at entry points to the islands as well as in formal government publications. Increases in the frequency of visits to the island by Parks Victoria staff, as well as pest plant and animal contractors, improved management visibility and supported relationships with the neighbouring properties and local businesses.

Poison baiting of Feral Pigs is widely practised across Australia (Choquenot *et al.* 1996) and was initiated on Quail Island in April 2016. At this time, the only registered poison bait product for pig control in Victoria was 1080 PIGOUT® (Animal Control Technologies Australia, Somerton). A Hoghopper™ feeder was installed on a trail used by Feral Pigs to reduce the exposure of non-target species to poisoned baits. Fermented grains and cracked wheat were initially provided as unpoisoned free-feeds prior to supplying 1080 PIGOUT™ baits.

Population estimates

An additional ten trail cameras (Snap, Faunatech, Bairnsdale) were installed in April 2019 to broaden the monitoring effort across the island in the lead-up to an aerial shooting program (Fig. 2). Cameras were established in horizontal orientation across trails, at the water

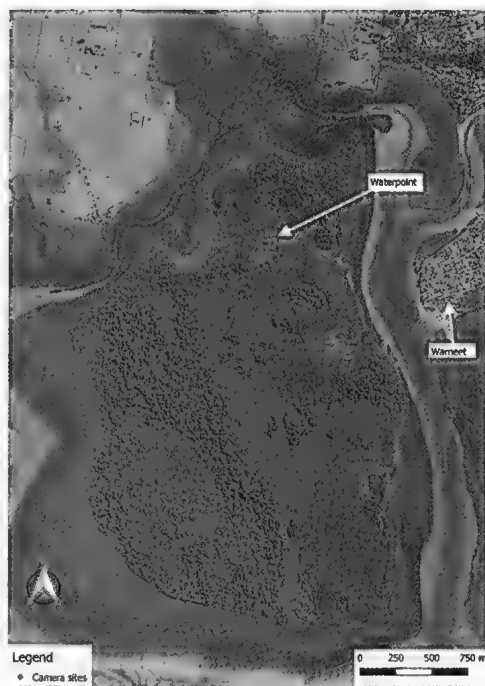


Fig. 2. Location of trail cameras used on Quail Island.

point and along the high tide line. Three photographs were taken per detection without a rest period. Cameras were serviced monthly with data being curated using CPW Photo Warehouse (Ivan and Newkirk 2016). A complementary project that was monitoring Southern Brown Bandicoot *Isodon obesulus* across the region shared copies of pig detections on the island and adjoining area (D Bryant, pers. comm. 21 July 2020).

Prior to the aerial shooting operation, a commercial drone pilot was engaged to fly a series of transects across the island with the objective of preparing an independent population estimate. The aircraft, a DJI M210 fitted with a Zenmuse XT-R Thermal Camera 30Hz 640x512 with a 19 mm lens, operated from Warneet boat ramp and flew pre-planned flight missions across ~500 m of Western Port before reaching Quail Island. Logistical issues associated with accessing the island at night through mangrove and mudflats with bulky equipment precluded operation of the drone from the island itself. The camera was configured to show heat-emit-

ting objects, such as animals, as white hot. Nine flights were undertaken on 12 May 2019 between 17:20–20:20 hrs at a maximum airspeed of 15 m/sec (Fig. 3). The aircraft was fitted with multiple high luminosity lights that ensured it was within the pilot's view at all times. The aircraft was flown at maximum altitude of 82 m AGL along east-west transects that were spaced 130 m apart. Video resolution was expected to be 24 cm per pixel at this altitude. Video recordings were reviewed simultaneously with replay of the flight log using the Google Earth flight simulator package.

Aerial shooting program

Preparation of documentation and approvals for the aerial shooting program began in February 2018. The choice of aircraft supplier and shooter was coordinated by the State Airdesk, with both operators endorsed with the Feral Animal Aerial Shooting Team (New South Wales Government). Aerial shooting operations were planned and delivered in accordance with Parks Victoria procedures, which included

engaging a veterinarian to observe and report on animal welfare aspects of the aerial shooting operation. Letter drops and signage at access points notified the surrounding community of the operation and provided a contact point for enquiries. Additional signage indicating the details of the park closure was erected the day prior to the operation, and notifications were given to Victoria Police and put on the Parks Victoria website.

The aerial shooting operation was undertaken on 23 May 2019, with low fog delaying take-off of the aircraft until 07:00 hrs. A Eurocopter AS350 Squirrel with doors removed was used. Crew onboard were limited to pilot, shooter and Parks Victoria observer. Aircraft crew communicated with each other and ground crews using voice-operated communications equipment. Feral Pigs were shot multiple times with a semi-automatic 12G shotgun using either size 00SG or AAA projectiles once the target was positively identified by observer and shooter. Aiming points were the head and/or thorax (Sharp 2012). Three flights were conducted by the aircraft in the area approved for aerial shooting. The first flight was terminated early due to fog at ground level, which precluded good visibility of targets. The operation then recommenced over Quail Island at 08:42 hrs. The aircraft systematically searched the island flying at ~30 m AGL for 121 minutes at slow speed, and operated a siren periodically to prompt the Feral Pigs to break cover.

Once the aircraft had left the site, a team walked to each carcass to collect morphometric measurements, undertake an assessment of shot placement for welfare reporting purposes, and take tissue samples for animal health and DNA identification purposes. A trail camera was installed to monitor which species scavenged on the carcasses.

Ground shooting

Following the aerial shooting operation, camera detections indicated a maximum of four individual Feral Pigs remained. Contract shooters were engaged to ground-shoot the remaining Feral Pigs, with their contract being paid out on the basis of achieving eradication of the population. Pig feeders prepared from PVC pipe were installed on 7 November 2019

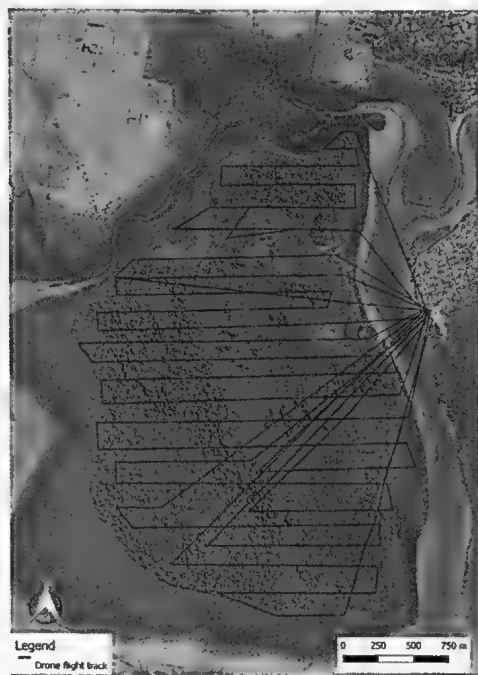


Fig. 3. Combined transects flown by drone with thermal sensor.

at two locations that were suitable for shooting and regularly used by Feral Pigs. The feeders were filled with a blend of fermented barley, corn and Carasweet Feral Pig attractant (Agricon Products, Maryborough, Queensland) and were serviced on 14 and 28 November 2019. Trail cameras with 4G phone telemetry (Swift Enduro, Outback Cameras Australia, Toowoomba, Queensland) were installed to monitor activity at the feeders remotely. A sow was regularly photographed feeding on Feral Pig carcasses after the aerial shooting operation, including during daylight hours. This individual was observed to be in poor physical condition and was last detected on 4 August 2019. It was apparent that only two adult boars remained during the feed training period.

These Feral Pigs first attended the feeder at the water point on 30 November 2019 and returned on the two subsequent nights. By dusk on 3 December 2019, two shooters established a position at this site and waited for the Feral Pigs to arrive. A thermal scope (Pulsar Trail XP50, Yukon Advanced Optics, Lithuania) fitted to each .308 calibre rifle enabled non-invasive observation of the Feral Pigs. Once the Feral Pigs were both settled and eating, the shooters simultaneously fired to ensure no opportunity for escape was available.

Results

Drone mounted thermal camera survey

Many 'hot-spots' were identified during video review, but the aircraft speed and altitude meant that these appeared on screen for <5 seconds as small white blobs with insufficient detail to make accurate determination of species. Koalas, possums and birds probably account for hot-spots that appeared to be in the tree canopy. It was possible to confidently ascribe only one detection to a Feral Pig, with this animal being located on the saltmarsh at the end of a transect where the camera panned while the aircraft hovered (Table 1). The combination of aircraft speed and wind encountered at altitude resulted in the endpoints of the camera gimbal being reached frequently, producing a jerky and disrupted field of view from different angles.

Aerial shooting

The aircraft was within the area approved for shooting for a total of 195 minutes (Table 2). A group of three similar-sized Feral Pigs was observed near the water point on the first flight. One of these individuals was shot (#1) but the fog prevented shooting of the others. It is probable that pig #2 was also one of these individuals seen (Fig. 4). The aircrew reported

Table 1. Thermal survey flights from a drone over Quail Island.

Flight No.	Time of flight (hrs)	Duration (min:sec)	Feral Pig Detections	
			High confidence	Low confidence
1	17:22–17:32	10:26	0	0
2	17:53–18:01	7:35	0	4
3	18:14–18:26	11:06	0	3
4	18:33–18:43	12:47	1	4
5	18:53–19:04	10:32	0	0
6	19:11–19:21	9:48	0	0
7	19:30–19:41	10:30	0	0
8	19:49–20:00	10:58	0	0
9	20:10–20:19	8:41	0	1

Table 2. Summary of aerial shooting results.

Flight No.	Time of flight	Feral Pig ID	Shot time (hrs)	Morphometric
1	07:07–07:21	1	07:20	Female, 46 kg
2	08:42–10:43	2	08:51	Male, 54 kg
2		3	09:26	Male, 74 kg
2		4	10:25	Male, 60 kg
3	13:15–14:05	-	-	-

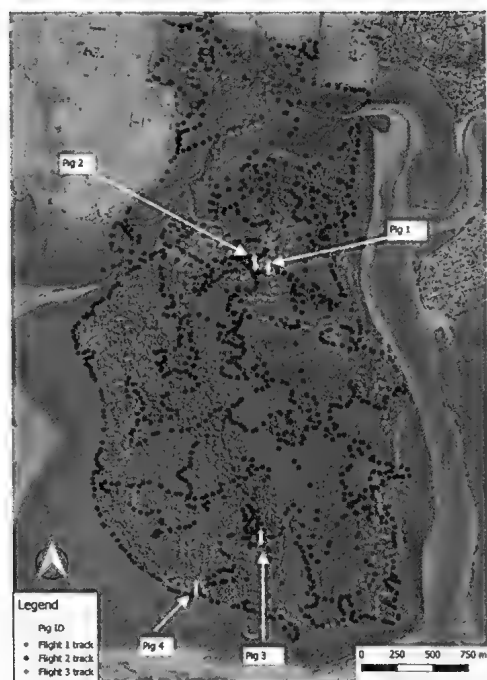


Fig. 4. Flight tracks of aerial shooting operation over Quail Island.

good visibility in flights 2 and 3 through the tree canopy, sufficient to also observe Black Wallabies *Wallabia bicolor* and European Red Foxes *Vulpes vulpes* on the ground.

Serology from the samples taken from these Feral Pigs produced negative results for zoonotic diseases including African swine fever, Q fever, brucellosis, influenza A and leptospirosis. Tissue samples have been preserved and contribute to a statewide Feral Pig biosecurity project.

Ground shooting

There was a delay of 10 and 23 days between installation and first visitation of Feral Pigs to the feeders at site 3 and the water point respectively. Both Feral Pigs had black pelage and were difficult to differentiate in the photos. They were photographed individually and together. Two male Feral Pigs, weighing 88 and 110 kg respectively, were shot at the water point feeders on 3 December 2019.

Camera monitoring

The time of camera detections was corrected for daylight saving time and activity patterns were calculated, allowing for a five-minute rest period between detections (Fig. 5). The trail cameras demonstrated that the Feral Pigs on Quail Island were predominantly nocturnal with the exception of the underweight sow, which was regularly photographed scavenging on pig carcasses following the aerial shoot. No physical injury was observed in the photographs of this individual, suggesting that it had not been wounded during the aerial operation (Fig. 6).

On a camera service on 5 May 2020, seven cameras were found to have been stolen. A remaining camera provided blurry photos of a human and two dogs on the evening of 6 March 2020. This was the first illegal activity observed on the island since April 2018.

No Feral Pigs were detected following the completion of ground shooting on 3 December 2019.

Discussion

Feral Pigs on Quail Island were causing significant ecological damage to soils and vegetation, and probably impacting populations of native fauna through direct and indirect processes. The introduction of Feral Pigs and subsequent hunting with dogs are both illegal activities within the conservation reserve. Eradication of the Feral Pig population was viewed as an achievable objective and the most appropriate option for Parks Victoria given the ecological significance of this site.

Several pest animal control contractors have been engaged to assist with Feral Pig control activities since these animals were introduced onto the island. Increased effort to eradicate the Feral Pig population was driven by allocation of a dedicated labour commitment supported by increased investment to protect the Ramsar Convention values and also coincided with an attempt to eradicate Feral Goats on an adjacent island.

The persistence of three Feral Pigs on Quail Island following the aerial shooting operation prompts questions about how these animals escaped detection from the aircraft. Given the

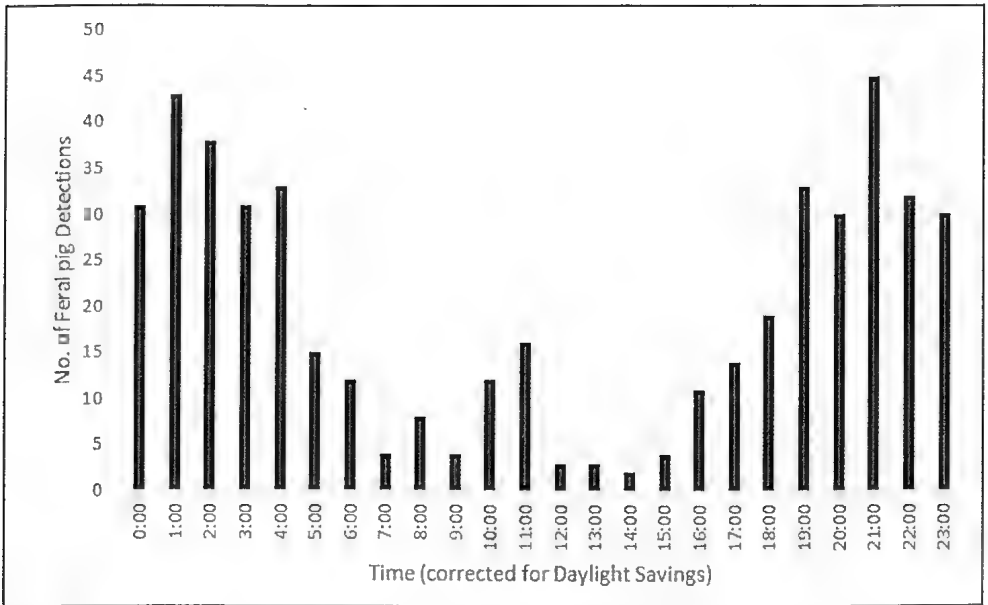


Fig. 5. Diel activity of Feral Pigs (May–December 2019) determined by detections by cameras with a five-minute rest period.



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Fig. 6. Chronically underweight female Feral Pig feeding on pig carcass on Quail Island.

body size, it is possible that the 'underweight sow' was the third individual sighted by the aerial shooting crew during the first flight. However, we do not know whether the two large boars avoided the helicopter by remaining in dense vegetation, or if they had left the approved shooting operation zone.

Land managers undertaking eradication operations on islands must be confident that they can detect and remove all individuals within the pest population (Bomford and O'Brien 1995; Clout and Veitch 2002). The increasing accessibility of thermal sensors in civil applications provides an additional tool that can improve the efficacy and accuracy of wildlife management operations (Lethbridge *et al.* 2020). Development and validation of these sensors in densely vegetated environments has been undertaken in New Zealand, where they have been incorporated into the Feral Pig aerial shooting strategy for the 57 000 ha Maukahuka Pest Free Auckland Islands project (Cox 2019). The observer searches for targets with a high-resolution thermal camera and then directs the pilot and shooter with a daylight visible laser. The firearm is fitted with a thermal scope and shots are fired only once the species has been confirmed by shooter and observer (Matthews in Parks Victoria 2019). This technology should be considered for aerial counts and shooting operations in densely vegetated sites within Australia.

Conversely, the drone-mounted thermal sensor was found to be inadequate when operated at the altitude and speed used over Quail Island. An unmanned aircraft was used for this count to reduce costs and disturbance of animals. Equivalent aircraft have been found to provide more accurate results for detecting Feral Pigs and similar-sized species at altitudes of ~50 m AGL, at ~5 m/sec ground speed using a forward-facing camera set at 55° (Johnson 2019; P Adams pers. comm. 30 July, 2020). Our pilot chose to operate at a higher altitude to ensure compliance with line of sight requirements and minimise the number of transects over Western Port. Variation in the camera direction during and between transects, as well as jerky vision associated with the gimbal reaching endpoints, contributed to the uncertainty about confirming the species detected

by the camera. In hindsight, a manned aircraft with a thermal sensor may have provided more effective pre- and post- survey counts given the size of Quail Island and logistics associated with accessing the island. The potential for disturbance of target and non-target species associated with low-level aircraft operation would have required consideration.

We encourage other land managers to consider inclusion of additional invasive species to the 'approved targets' in the planning process for aerial shooting operations to fully capitalise on the opportunity afforded by the aircraft platform where the eradication objective is not likely to be compromised. In this case, two adult European Red Foxes were observed by the shooter, but no shots were fired as this species was not listed on the shoot plan (D Rogers, pers. comm. 25 May 2020).

The removal of Feral Pigs on Quail Island is estimated to have cost ~\$200 000 since 2015, with this figure including agency and contractor labour as well as operational expenses during the monitoring and eradication stages. Considered in isolation from other projects (e.g. the Feral Goat project), the combination of modern thermal optics and ground shooting by professional operators may have been sufficient to achieve the eradication outcome at reduced cost when compared to the aerial platform used at this site. Ongoing expenditure will be incurred to maintain the surveillance effort to secure the investment made to date.

Invasive species eradication projects on islands typically include considerations of the probability of achieving eradication success based on the behaviour of the target species, survey effort and size of the island (Parkes *et al.* 2010; Ramsey *et al.* 2009). The size of Quail Island and detectability of Feral Pigs on the island suggest that the population was eradicated in December 2019. Correlative observations of Southern Brown Bandicoot digs on the banks of the water point during autumn 2020, the first in six years, as well as recovery of vegetation along trails and saltmarsh, also suggest that Feral Pigs have not been active since this time.

However, we note that the illegal activity, including presence of dogs and theft of cameras that occurred in March 2020, was the first in ~23 months. This may be an ongoing problem,

given the proximity of the island to a major city, but the ecological significance of the site warrants ongoing vigilance. The experience and lessons from the eradication operation have improved expertise and skills within Parks Victoria staff and contractors who continue a surveillance effort and contribute to a rapid response plan to deal with the potential for incursions of Feral Pigs into the future.

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When Cats turn bushranger: a case study of policy and planning a pest eradication

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Abstract

Feral Cats *Felis catus* impact native wildlife species through direct predation and transmission of parasites. In July 2018, the Government of Victoria declared Feral Cats to be a pest in specific Crown land tenures and subsequently made changes to regulations to provide opportunity for use of additional control tools. This coincided with an Australian Government initiative to create *Feral Free Safe Havens* on islands and fenced enclosures. French Island was one of the islands nominated in this strategy and has had over 1100 Feral Cats removed since 2010. In conjunction, there has been increased awareness throughout the island community about the responsibilities of Domestic Cat owners. This paper presents the current regulatory environment and discusses how it impacts on a proposal to eradicate the Feral Cat population on French Island, Victoria. (*The Victorian Naturalist* 137(6), 2020, 228–239)

Feral Cats *Felis catus* were introduced into Australia at multiple coastal locations between 1824 and 1886 (Abbott 2002, 2008). Since these introductions, they have established populations across the entire continent with islands and predator-fenced reserves forming the only Feral Cat-free areas (Rolls 1969; Legge *et al.* 2017). The impacts of Feral Cat predation on Australian native fauna are widely reported (Dickman 1996; Hilmer *et al.* 2010a; Doherty *et al.* 2015, 2017; Woinarski *et al.* 2017a; 2017b; 2018) and there is increasing concern about the impact of cat-mediated disease transmission on both wildlife and agricultural livestock (Groenewegen 2018; Taggart *et al.* 2019). Removal of Feral Cat populations from islands is a pragmatic solution to reduce the threats to insular fauna (Twyford *et al.* 2000; Campbell *et al.* 2011; Parkes *et al.* 2014; Hanson *et al.* 2015; Robinson *et al.* 2015).

In his recollections of hunting throughout the Greater Melbourne area during the 1850s, Wheelwright (1862, p. 49) noted that the 'domestic cat sometimes wanders ... and turns bushranger'. Around the same time, a popu-

lation of Feral Cats was reported on French Island, following intentional releases by pioneer farmers (Lewis 1934; Gooch 2006). Since then, there has been recruitment from straying Domestic Cats. Feral and Domestic Cats negatively impact native species on French Island by direct predation (Fig. 1) and by spreading the disease toxoplasmosis into the environment (McTier 2000; Johnston, pers. obs. since 2004; Groenewegen 2018). The Office of the Threatened Species Commissioner included French Island as one of five Australian islands to become Feral Free Safe Havens for wildlife species through eradication of Feral Cat populations (Department of Environment and Energy 2019).

Prior to December 2019, Victorian conservation land managers undertaking Feral Cat control were able to set cage traps and were required to transport any cats caught (Feral or Domestic) to an animal shelter. Seizure, including field shooting of cats, was also supported for authorised officers under the *Wildlife Act 1975* and *National Parks Act 1975*.

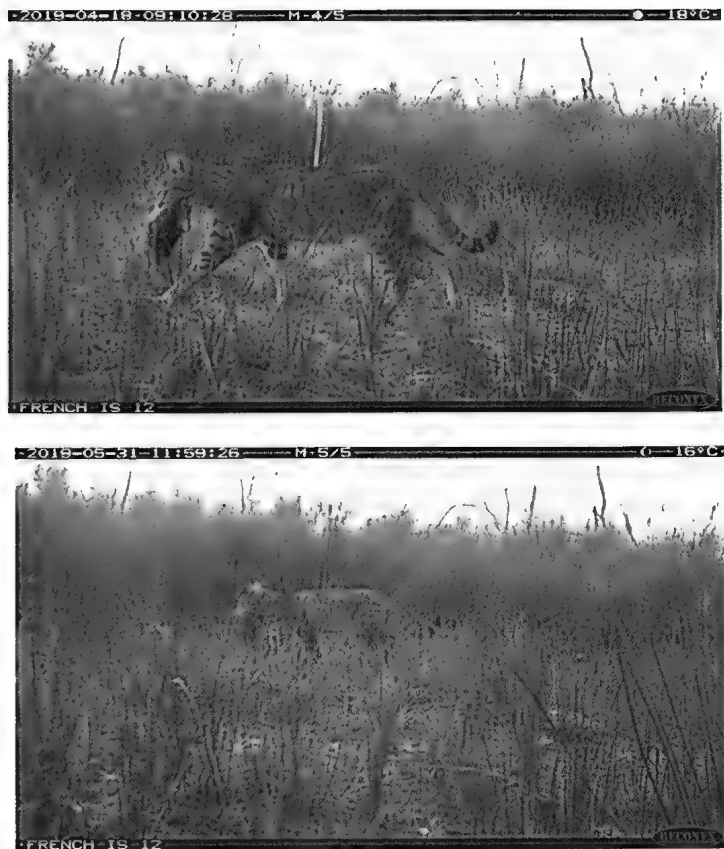


Fig. 1. Trail camera detections showing the same Feral Cat preying on two Lewin's Rail near Clump Lagoon, French Island National Park. Upper: 18 April 2019; Lower: 31 May 2019.

Key planning documents such as Victoria's *Biodiversity 2037* (Department of Environment, Land, Water and Planning [DELWP] 2017) recognise that biodiversity has an intrinsic right to exist, stress the importance of biodiversity protection on private land and set targets for control of pest predators such as Feral Cats. The *Inquiry into the control of invasive animals on Crown Land* found that 'current Victorian legislation prevents any effective control of Feral Cats' and recommended (p. 206):

'That the Government declare feral or wild cats to be 'established pest animals' under the *Catchment and Land Protection (CaLP) Act 1994*, mirroring the way wild dogs are classified'.

(Parliament of Victoria, Environment, Natural Resources and Regional Development Committee 2017).

In response, the Government of Victoria provided a definition for Feral Cats in July 2018 and declared these animals as a pest species under the *CaLP Act 1994*, which required that these species 'should be eradicated or controlled or its spread in the wild should be prevented' (Part 8, section 67, p. 107). Feral Cats are 'unowned and live completely independently of humans with respect to food and shelter and without veterinary care. Feral Cats survive and reproduce in self-perpetuating populations in the wild' (DEWLP 2018, p. 1). Feral Cats are 'widespread and can be found in all terrestrial habitats in Victoria' and they 'have a major impact on Victoria's biodiversity and are one of the most significant threats to our native wildlife' (DELWP 2018, p. 1). Where the declaration takes effect (i.e. on land managed by

Parks Victoria, DELWP, Phillip Island Nature Parks and four alpine resorts), the declaration requires that managers of Crown land undertake control of Feral Cat populations 'targeted to protecting the threatened wildlife most at risk of predation by Feral Cats' (DELWP 2018, p. 1). The code of practice for Feral Cat control on Victorian Crown land provides information on the policy and legislative framework within which control is undertaken while also seeking to 'limit potential negative effects of control on native species, including unexpected consequences, to prioritise humane control methods that minimise animal suffering and to address risk posed to domestic cats' (Agriculture Victoria 2020a, p. 2).

Following the Feral Cat declaration, there has been no change to regulations applying to land managers intending to undertake Cat control outside of the declaration areas (Fig. 2). However, following amendment of the *Agricultural and Veterinary Chemicals (Control*

of Use) Act 1992, management of populations can now include use of poisons containing para-aminopropiophenone (PAPP), but not sodium fluoroacetate (1080) (Victoria Government Gazette 2019). This policy acts to prevent operational use of the Felixer grooming trap (Read *et al.* 2019) in Victoria as the device currently delivers a toxic gel containing 1080. Felixer has, however, been used on Phillip and French Islands in the non-firing camera-only mode to collect local efficacy data to support future operations.

The *Prevention of Cruelty to Animals Regulations 2019* now permits use of rubber-padded leghold traps for capture of Feral Cats, but only within specified declaration areas. Proposals to use leghold traps require Ministerial approval and are limited to sites where eradication of populations is achievable, i.e. islands where it has satisfactorily been demonstrated that alternative tools are insufficient to achieve this objective. Consideration may also be

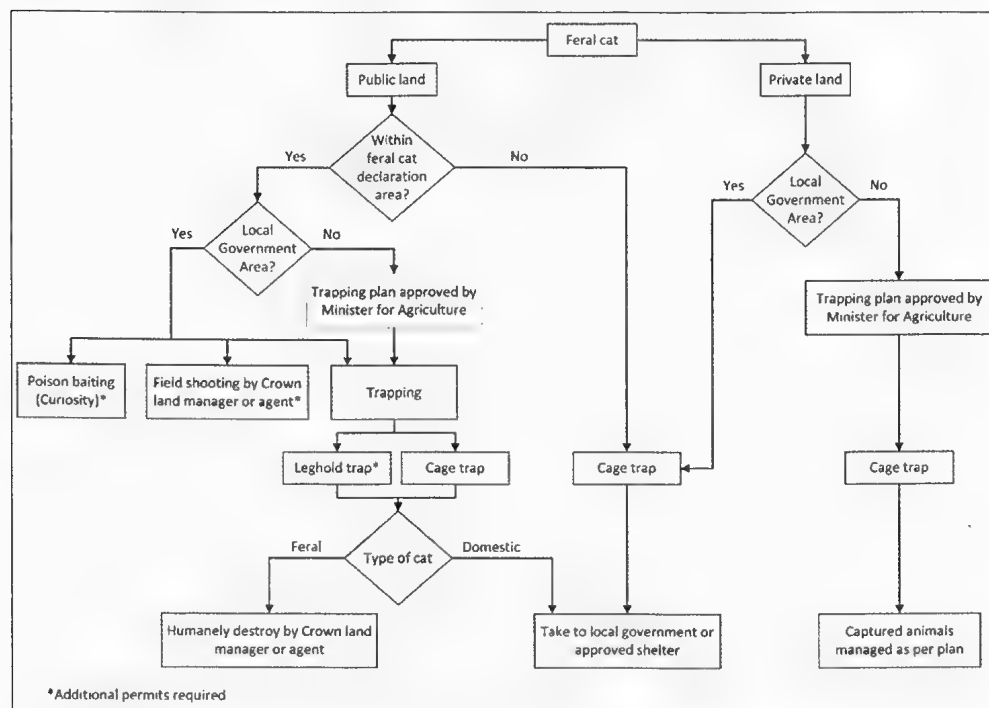


Fig. 2. Logic diagram indicating tools and techniques available for cat management in Victoria.

given to approve use of the traps for short periods of time following declared emergencies (e.g. bushfires) where localised populations of threatened or endangered species are at risk of heightened predation from Feral Cats. In comparison, European Red Foxes *Vulpes vulpes*, which also are a declared pest species found across the state (other than for offshore islands), may be captured using the same rubber-padded leghold traps on both private land (other than within populous areas) and Crown land following receipt of Ministerial approval. Leghold traps may also be set throughout defined areas of the state on both private and Crown land estate to capture wild dogs, but their use on Crown land is typically limited to the area within three kilometres of the freehold boundary (Agriculture Victoria 2020b).

Approved procedures for the ground shooting of Feral Cats, including the euthanasia of trapped animals, have been published (Parks Victoria 2020). The use of trained scent dogs is supported. This is expected to assist in the detection and removal of Feral Cats in research projects as well as eradication operations.

Exclusion fencing is a non-lethal control tool used at several sites within Victoria, on both public and private land, to prevent invasive species from impacting high-value populations of threatened species, e.g. at Woodlands Historic Park, Mt Rothwell, Tiverton and Neds Corner Station. Materials and labour costs for effective exclusion fence construction is typically between \$40 000 and \$50 000 per kilometre with an ongoing requirement for investment into maintenance and incursion surveillance (A Rypalski, pers. comm. 19 September 2020). Once construction is complete, the removal of resident predators from within the 'mainland island' then requires a range of strategies to complete eradication.

Within Victoria, the *Domestic Animals Act 1994* complicates Feral Cat control operations as it does not require Domestic Cats to be contained on their owner's property, unless this is required by a local government bylaw. This has resulted in an inconsistent approach across the state. In some areas, it effectively allows for Domestic Cats to enter areas where Feral Cats must be controlled. In contrast, the Act requires domestic dogs to be contained to their owner's

property, or otherwise remain under control. The *Domestic Animals Act 1994* provides limited powers for landowners and their agents to shoot cats that threaten contained or tethered livestock.

While most of Victoria falls within a Local Government Area, there are a small number of 'Unincorporated Localities' referenced in the *Planning and Environment Act 1987* that do not have local government. Instead, planning issues in Unincorporated Localities, such as the alpine resorts and French Island, are considered by DELWP through the office of the Minister for Planning. Alpine Resort Boards are referral authorities that have powers to manage domestic animals under the *Alpine Resorts (Management) Act 1997*. The requirements of the *Domestic Animals Act 1994*, such as registration and identification of Domestic Cats as well as permitting seizure and levying of fines on owners, are directly conferred to local government. In the absence of this tier of government, it is probable that the requirements of the *Domestic Animals Act 1994* are not enforceable in Unincorporated Localities, which presents a problem given the high conservation significance of these locations. The recent regulatory changes have included a requirement for the Minister for Agriculture to approve a plan for domestic animal management prior to initiation of rubber-padded leghold trapping in Unincorporated Localities.

Feral Cat management on French Island

The revisions to the Government of Victoria regulation have coincided with a proposal to eradicate Feral Cats on French Island. Centred within the internationally significant Western Port Ramsar site, French Island (38°21'S, 145°21'E) covers an area of 17 000 ha and is highly significant for biodiversity conservation in Victoria. Approximately two-thirds of the island, an area of 11 100 ha, is managed as the French Island National Park (Fig. 3). The island has extensive mangrove and saltmarsh areas that transition to heath and open pasture in the interior. Wetlands are extensive and diverse across the island, ranging from seasonal to permanent open water, providing habitat for an array of flora and fauna (Weir and Heislors 1998; Quinn and Lacey 1999; Lacey 2008).

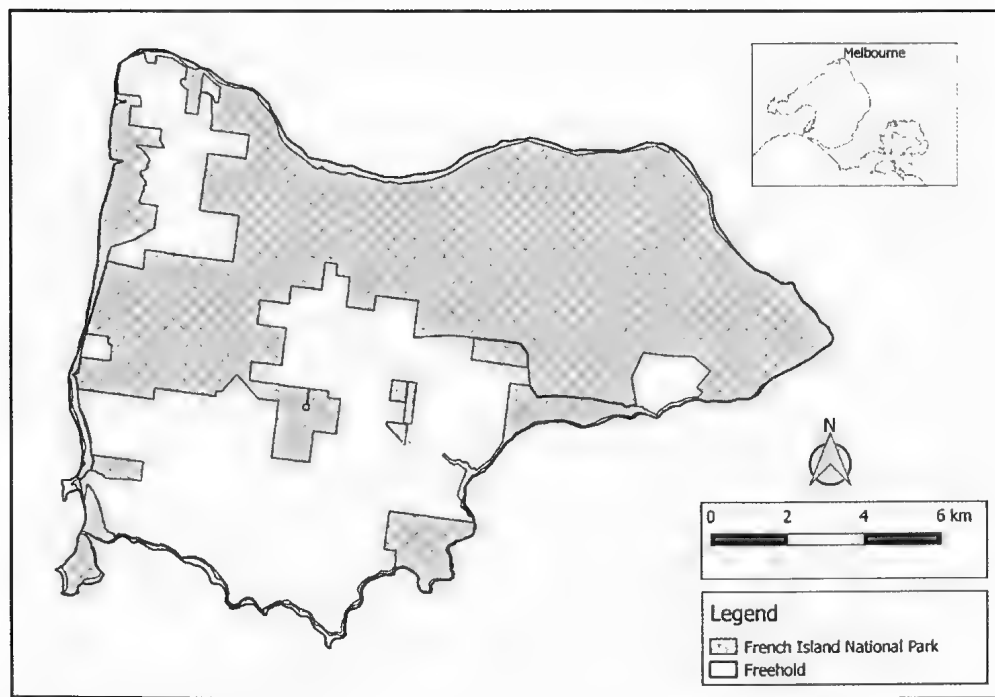


Fig. 3. Land tenures on French Island.

More than 230 bird species have been recorded on the island, with 32 of these being listed as Threatened under state and/or federal environmental classifications, including King Quail *Excalfactoria chinensis victoriae*, Orange-bellied Parrot *Neophema chrysogaster*, Fairy Tern *Sternula nereis* and Lewin's Rail *Lewinia pectoralis pectoralis* (Quinn and Lacey 1999; DELWP 2019). French Island has a limited native mammal and reptile assemblage in comparison to the adjoining mainland (Kirkwood and Johnston 2006), but includes significant populations of Long-nosed Potoroo *Potorous tridactylus tridactylus* and Swamp Skink *Lissolepis coventryi*. Zoos Victoria translocated 74 adult Eastern Barred Bandicoots *Perameles gunnii* onto the island in 2019, with evidence of successful breeding subsequently observed (A Coetsee, pers. comm. 2020). Orange-bellied Parrots released at an adjacent mainland location in 2020 are expected to forage on French Island's salt-marshes (A Herrod, pers. comm. 2020).

The Australian Bureau of Statistics (2016) reported 119 residents living on freehold

properties with absentee landowners also owning property on the island. At June 2020, there are 37 Domestic Cats residing on 22 properties on the island. Thirty-five of these Cats have been sterilised. Domestic Cats have been trapped and returned to their owners on at least 40 occasions during the annual cage trapping programs between 2010 and 2020. Roads and adjacent areas on the island are managed by DELWP, which facilitates the trapping of Feral Cats in these areas.

Cage trapping

Prior to 2008, Parks Victoria undertook strategic cage trapping operations to remove Feral Cats to temporarily reduce predation pressure on migratory birds (M Douglas, pers. comm. 2 May 2000). In 2008, broader-scale control within the National Park was undertaken as part of the development of a poison bait development (Johnston *et al.* 2011). In June–July 2010 and 2011, Parks Victoria undertook eight-week cage trapping programs within their estate (Norvick 2015). This effort was then extended

in 2012 with trapping also undertaken over an additional eight weeks (May and August) in the freehold area of the island by the French Island Landcare Group (Landcare). The complementary trapping effort has continued to the current day and uses up to 100 cage traps (800 × 310 × 280 mm, P&L Wire Products, Frankston, Australia) per night baited with fried chicken pieces. Cage traps are typically set along vehicle access tracks to simplify daily servicing and are operated continuously unless poor weather conditions are forecast. Landcare maintains a photographic database of pet cats on the island to support the rapid return of pets caught during the trapping program. Trapped cats are identified as being 'domestic' or 'feral', referencing the database and behaviour, with the Feral Cats being euthanased by shooting using a .22 calibre rifle. The Australian Government has fully subsidised a voluntary program to sterilise Domestic Cats on the island, preventing unwanted litters that could otherwise contribute to the feral population. Passive Integrated Transponder tags are also implanted by the veterinarian.

Population monitoring of Feral Cats and wildlife

An island-wide grid of trail cameras (Reconyx HF2X, Holmen, WI, USA) was established in August 2018 such that there was a camera within every ~2 km² cell. Cameras were mounted horizontally at a height of 400 mm and were aimed across a vehicle track. The data from these cameras informs occupancy and abundance estimates of Feral Cats and native wildlife. Ground-dwelling birds were also monitored using acoustic monitors, trail cameras and call playback surveys (Znidarsic 2017). These species are highly susceptible to cat predation given their size as well as nesting and foraging behaviours (Woinarski *et al.* 2017b; Geyle *et al.* 2018).

Cameras are serviced quarterly with photographic data screened to remove the false detections. Photos are then imported into a software package—Photo Warehouse (Ivan and Newkirk 2016)—and wildlife classified to species level. Estimation of the Feral Cat population size was made using Spatial-Mark-Resight

models from the *secr* package in R (V3.4.4) (University of Otago 2018; R Core Team 2018). A photographic catalogue of Feral Cat images was prepared from the camera data and a unique identifier was ascribed to cats with distinct natural markings that could be confidently recognised from the images. This permitted a detection history to be prepared using cat identity, geographic location and time. Cats that were not identifiable due to pelage (black, ginger) or motion blur were recorded as 'unmarked'. Detection histories for recognisable individuals at each camera site, as well as detections of unmarked individuals contributed to statistical models, were used to generate the population estimate.

Feral Cat ranging behaviour

Ten Feral Cats were trapped within the National Park and fitted with VHF/GPS datalogger collars (Sirtrack, Havelock North, NZ) in February 2008. Locations were recorded at three-hourly intervals with data subsequently filtered to remove points that had a horizontal dilution of precision that was >5.0. The *ade-habitatHR* package (R Core Team 2019) was used to estimate 95% kernel daily ranges. For cats using coastal areas, the resulting polygon was clipped to the island coastline to correct the home range area estimations.

Results

Cage trapping

Approximately 200 cats were removed as part of cage-trapping programs conducted prior to 2010 (McTier 2000; Marks *et al.* 2006; Johnston *et al.* 2007; Hilmer *et al.* 2010b; Johnston *et al.* 2011) with recreational shooters also frequently removing cats.

A total of 1161 Feral Cats were trapped between 2010 and 2020 over approximately 85 331 cage trap nights (39–385 cats per year). Mean capture efficacy was 0.9 cats/100 trap nights (range 0.4–1.7). The estimate of capture efficacy is conservative due to accurate records not being maintained.

Most cats trapped were classed as adults (Fig. 4a). There was no sex bias in captures per year (Chi² *p* values 0.09–0.87) or across all years (Chi² *p* value 0.33) (Fig. 4b).

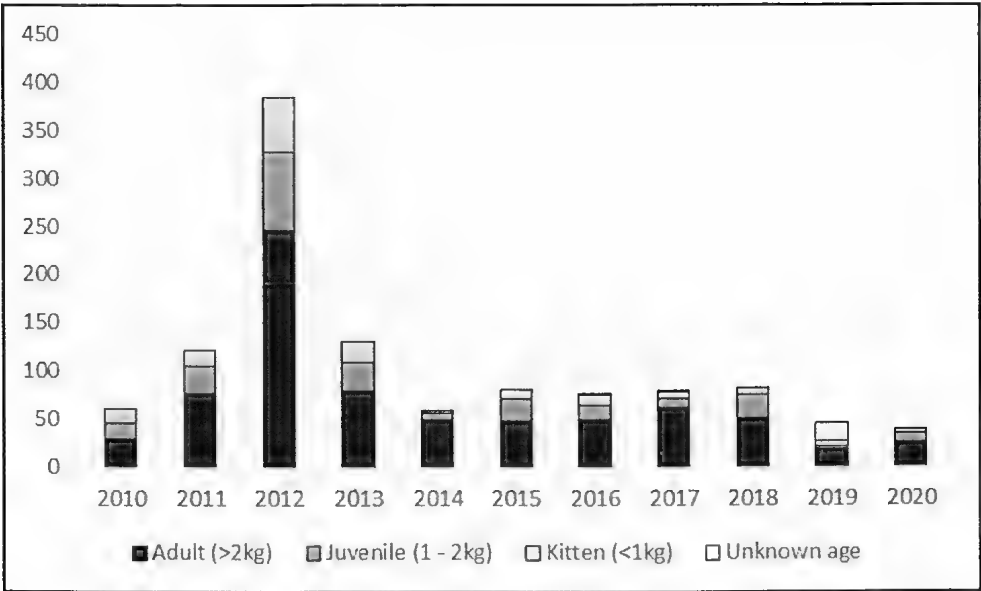


Fig. 4a. Age class of Feral Cats trapped on French Island 2010–2020.

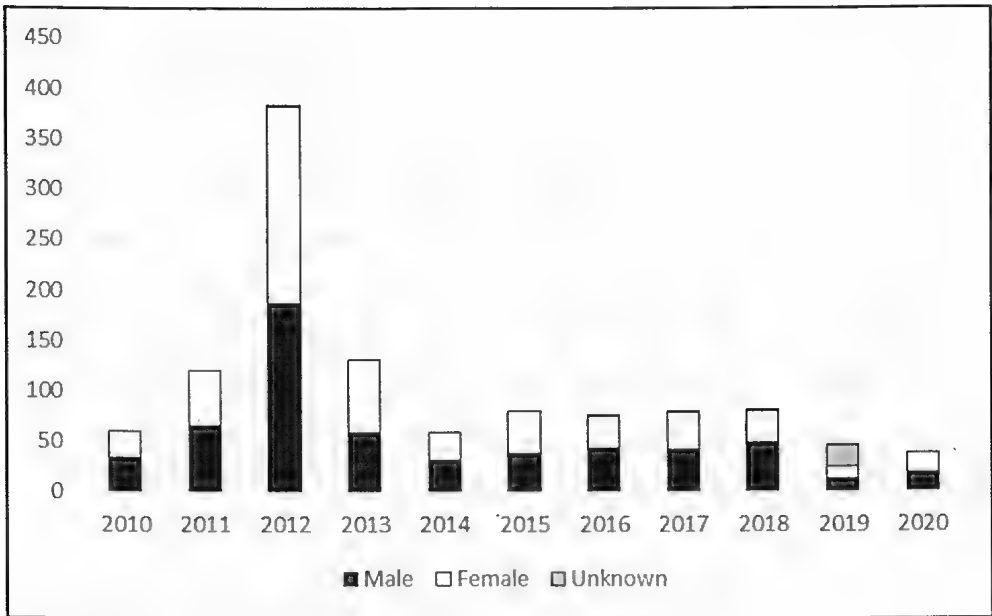


Fig. 4b. Sex of Feral Cats trapped on French Island between 2010 and 2020.

Population monitoring of Feral Cats and wild-life

Cats were detected at 46 of the 59 camera sites across the island over a period of 88 days between 17 August and 12 November 2018. The average density of Feral Cats on the island was estimated to be 0.56 cats/km² (95% confidence interval: 0.43–0.71). The total abundance of Feral Cats on the island was estimated to be 97 (95% confidence interval: 76–124).

Feral Cat ranging behaviour

Daily activity ranges for the Feral Cats fitted with GPS dataloggers in 2008 ranged from 2.4–10.2 km² (Table 1). Long-distance ranging behaviour was observed in one male Feral Cat that completed a ~58 km circuit covering much of the island over a period of 17 days (Fig. 5). Another adult male Feral Cat spent 74 consecutive days ranging exclusively along the northern saltmarsh.

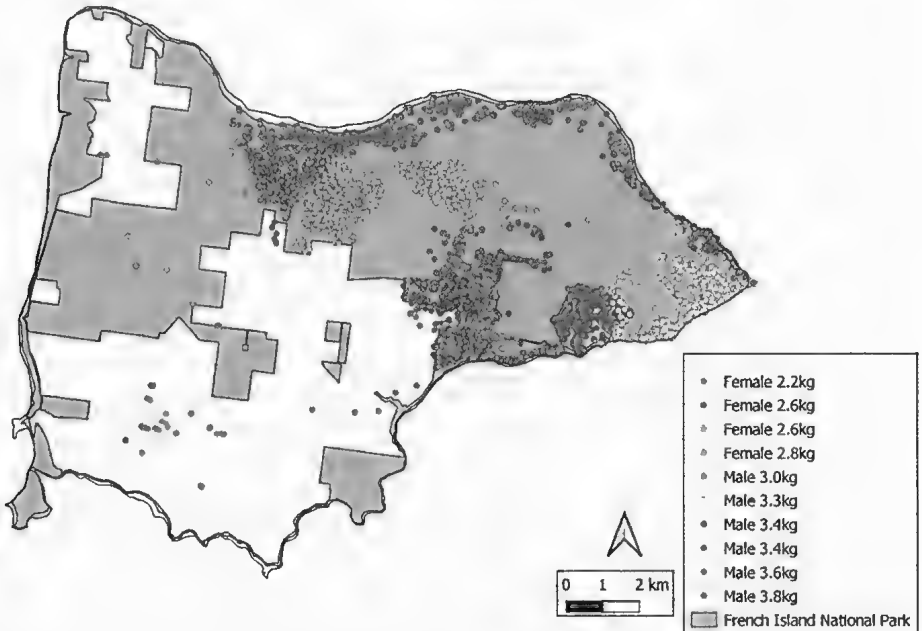


Fig. 5. GPS activity data from individual Feral Cats on French Island in 2008.

Table 1. Morphometric details, 95% home range utilisation distribution and mean daily distance travelled by GPS-collared Feral Cats on French Island in 2008. M=male; F=female; sd=standard deviation; min=minimum distance travelled; max=maximum distance travelled.

Cat ID	Sex, weight (kg)	Kernel 95% (km ²)	Mean daily distance (km)	sd	min	max	Dates active	Data points
0400	M, 3.0	10.2	1.9	1.4	0.1	5.5	9 February – 1 May	82
1200	M, 3.4	2.6	2.9	1.1	0.1	6.4	7 February – 27 May	109
1400	M, 3.6	6.2	3.4	2.1	0.0	9.3	7 February – 21 May	102
1600	M, 3.3	4.6	3.3	1.8	0.3	7.8	8 February – 7 June	121
2400	F, 2.2	2.9	0.9	0.5	0.0	2.1	8 February – 16 May	64
2600	F, 2.6	2.4	1.4	0.9	0.0	3.9	14 February – 14 May	87
3000	F, 2.6	4.9	2.6	3.3	0.0	18.3	28 February – 10 May	78
3600	F, 2.8	2.7	1.5	0.8	0.0	3.1	7 February – 23 Apr	19
3800	M, 3.4	7.2	1.1	1.3	0.0	5.5	7 February – 27 June	135

Discussion

Our study shows that the intensive use of cage traps over a decade was insufficient to achieve effective control of the Feral Cat population on French Island. This population is self-sustaining, being independent of Domestic Cats on the island and is not subject to immigration of individuals from elsewhere. While many Feral Cats are trapped annually, sufficient breeding age animals do not enter the cage traps and sustain the feral population (Johnston *et al.* 2019a). The persistence of these 'untrapped' cats is now being documented across the island at camera sites. It is clear that eradication success will never be achieved if this management approach is to continue.

Capture success has been largely stable over the program, albeit with some variation in 2012 and 2019, with equal proportions of males and females and most being of breeding age. The extraordinary number of captures in 2012 is thought to be related to increased presence of House Mouse *Mus musculus* post-drought, with dead mice frequently found under the footplate in traps that caught cats (D Stephenson, pers. obs. 2012). A reduction in trapper effort contributed to the lower result in 2019. The cage trapping program has effectively resulted in an annual 'harvest' of Feral Cats, which we consider to be an unfortunate and avoidable outcome given the sustained impact on wildlife and cat populations it has created.

Recent regulatory changes in Victoria have provided an opportunity to equip some Crown land managers with more effective tools to reduce Feral Cat impacts on wildlife populations, but made no provision for these tools to be used on private land, even when the land is managed explicitly for biodiversity conservation. It is premature to determine whether these tools and, more specifically, the policies that enable their use, will improve control of Feral Cat populations across the state compared to the situation prior to the declaration. The guidelines permitting the use of leghold traps for capture of Feral Cats restrict the use of these traps only to sites where 'eradication' can be achieved, i.e. French and Phillip Islands, and do not appear to extend the opportunity for their use in mainland conservation reserves other than for emergency situations. Further, the Victorian policy

does not include support for the use of leghold traps in scientific research, as exists in other states (DAWE 2020). Modern rubber-padded leghold traps provide a more effective capture tool than cage traps and have been successfully used in capture and release studies (Campbell *et al.* 2011; Johnston *et al.* 2011; Algar *et al.* 2020). While there appears to be a reasonable prospect of receiving Ministerial endorsement to use leghold traps on French Island, the policy that governs the use of this tool will frustrate Feral Cat control and research programs elsewhere.

As of September 2020, the procedures governing the use of poison baits for Feral Cats have not been made available to agencies included within the Feral Cat declaration. DELWP have provided advice suggesting that these procedures will be completed in time to support use of the Curiosity* bait within French Island National Park in 2021. The proposed area and pattern used in this operation will largely replicate that used in the 2008 field efficacy study, with application timed to minimise hazard to wildlife species (Johnston *et al.* 2011; 2019b). Shooting will only be able to be used to a limited extent in the eradication project given the density of vegetation within the National Park and constraints associated with the *Domestic Animals Act 1994*.

The activity data derived from GPS collars fitted to Feral Cats on French Island demonstrate the extent to which individuals inhabit areas distant from vehicle tracks where they would infrequently encounter traps. A current study, also using GPS collars, has demonstrated the extent to which Feral Cats access private land on the island (Johnston *et al.* unpublished data). Application of control tools in these inaccessible areas is essential to put all Feral Cats at risk and satisfy the established criteria for successful eradication programs (Parkes 1990; Bomford and O'Brien 1995). Aerial distribution of toxic baits will greatly improve the encounter-opportunity for Feral Cats that infrequently access vehicle tracks. This will be important to remove female cats that typically maintain small activity ranges on French Island.

Anecdotal observations suggest that the existing Feral Cat control program has been worthwhile given increases in the number of Purple

Swamphens *Porphyrio melanotus* and Long-nosed Potoroos (D Stephenson, pers. obs. over 2019). Our trail camera and acoustic data has also confirmed the presence and persistence of ground-dwelling bird species that will provide a valuable dataset that can benchmarked against in future analyses. The cameras have also provided insights into Feral Cat behaviour, including stalking and carrying prey items, with one individual cat twice observed carrying a dead Lewin's Rail (Fig. 1).

Community attitudes to the management of cats on French Island have changed over the term of the project, with most Domestic Cats on the island now sterilised. This transformation was led by Parks Victoria, Landcare and members of the local community that progressively sought to foster Responsible Pet Ownership principles on the island. The subsidy provided for sterilisation and rapid return of trapped Domestic Cats to their owners is likely a key part of this attitudinal change. There is a recognition that landowners will continue to own Domestic Cats on French Island into the future and that this will not impact on the success of the eradication project provided all Domestic Cats are sterilised. This strategy differs from typical Feral Cat eradication programs on islands that commonly grant a 'last cat' exemption to owners, and recognises the proximity of French Island to the mainland (Algar *et al.* 2011).

Conclusion

There is a tension between the Government of Victoria's definition of a Feral Cat and understanding of the threat that these 'bushrangers' represent in the policies that govern use of effective tools to manage these populations.

A unified and collaborative approach to policy development and effective application of tools will facilitate the eradication of Feral Cats on French Island and improve predator management outcomes on conservation land more broadly in Victoria. A successful eradication outcome on French Island would bring an end to the annual killing of Feral Cats, allow island residents to keep sterilised Domestic Cats and create a safer haven for wildlife.

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Managing willow invasion in alpine Victoria

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Abstract

Willow *Salix* species are now recognised as posing a significant threat to the hydrological and biological values of rivers and alpine peatlands in eastern Victoria. Willows produce many tiny seeds that are able to disperse long distances, but they are short-lived and need moist open substrates on which to establish. The perfect conditions for willow seedling establishment in alpine and sub-alpine peatlands were created by the alpine fires in 2003, resulting in widespread mass germination of willows in 2004. This paper describes the subsequent research and management efforts undertaken to control these willows and so limit damage to these important alpine plant communities and habitats. (*The Victorian Naturalist* 137(6), 2020, 239–245)

Keywords: *Salix cinerea*, environmental weeds, seedling and seed source control, threatened alpine peatlands

Introduction

Willows were brought to Australia by European colonists. They were planted widely during the 1860s to 1980s for soil stabilisation along waterways and streams following vegetation clearance and for river improvement works, but they have become a serious environmental weed, spreading in natural waterways. Willows disrupt aquatic ecosystems, out-compete native vegetation, reduce water quality, block waterways and use more water than native riparian vegetation (Serena *et al.* 2001; Greenwood *et al.* 2004; Holland-Clift and Davies 2007; Doody and Benyon 2011). They are also a threat to the biodiversity of alpine peatlands (Threatened Species Scientific Committee 2008). In 2000, all willow taxa (except for *Salix babylonica*, *S. x calodendron* and *S. x reichardtii*) were classified as Weeds of National Significance (Australian Government 2020).

Salix is a widespread genus of shrubs and trees native to the cold and temperate regions of Europe, Asia and North America. Willows fall into two broad taxonomic groupings (Cremer 2003): tree willows (subgenus *Salix*), which include Weeping Willow *S. babylonica*, Black Willow *S. nigra*, White Willow *S. alba* and Crack Willow *S. fragilis*; and shrub willows (subgenus *Vetrix*), which include Grey Sallow *S. cinerea*, Common Osier *S. viminalis* and Purple Osier *S. purpurea*. Species from the same subgenus readily hybridise and there are a number of well-established hybrid taxa, including Hybrid Crack Willow or Basket Willow *S. x rubens*, a hybrid of *S. alba* and *S. fragilis*, and two hybrids of *S. cinerea*: Pussy Willow *S. x calodendron* = *S. cinerea* × *caprea* × *viminalis* and Sterile Pussy Willow *S. reichardtii* = *S. cinerea* × *caprea*. Only one sex of most taxa was introduced to

Australia and so these taxa do not produce seeds. *Salix cinerea* and *S. nigra* are the only two species recognised as producing seeds in Australia. However, species readily hybridise in the wild if their flowering times and distributions overlap, and there are now many undescribed hybrids in Australia that likely produce seeds (Cremer 2003).

Willow species produce millions of small seeds capable of dispersing many kilometres via wind (Fig. 1) (Hopley and Young 2015). In addition to producing seeds, *S. cinerea* and *S. nigra* are also able to colonise non-saline wetlands in addition to riparian zones, rendering these the most invasive willows in Australia (Cremer *et al.* 1995). Long-distance dispersal means that these species can spread across catchments and along waterways (Hopley and Young 2015). *Salix cinerea* is a particular problem in eastern

Victoria as it is more widespread than *S. nigra* and has spread into alpine vegetation, including into endangered alpine *Sphagnum* bogs and fen plant communities (aka alpine peatlands) (Cremer 2003; Moore and Runge 2012; Hopley and Young 2015). These alpine peatlands are listed as a threatened community under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (Alpine *Sphagnum* bogs and associated fens) and Victorian *Flora and Fauna Guarantee Act 1988*. *Salix cinerea* was widely planted in NE Victoria from the 1950s to 1980s for stream bank stabilisation as part of programs undertaken by river improvement trusts (North East Catchment Management Authority [NECMA] 2003). This included plantings around Rocky Valley Dam and the headwaters of Rocky Valley Creek and Pretty Valley Creek following construction



Fig. 1. Top left: an unburnt alpine peatland on the Bogong High Plains in Victoria. Top right: a burnt alpine peatland in 2010 with willows that established after the 2003 fires. Bottom left: a mountain stream with a dense infestation of mature willows in 2007, which may have contributed seed to the alpine infestation. Bottom right: a team of contractors removing willow seedlings on the Bogong High Plains in 2008.

of the Kiewa Hydro Electric Scheme from the 1930s to 1960.

Since the 1990s, *Salix cinerea* has caused problems and been managed in waterways in eastern Victoria (NECMA 2003). An assessment of willow establishment in the Australian Alps National Parks identified mature seed-producing *S. cinerea* on the Bogong High Plains, noted the potential for post-fire invasion and recommended removal of these plants (Carr *et al.* 1994). The 2003 fires created ideal conditions (moist open substrates) for *S. cinerea* seedling germination. Mature *S. cinerea* was removed from Falls Creek and Rocky Valley Dam in 2003 (Parks Victoria 2014) but there was still a lot of mature willow in the region. Mass germination of seed on the Bogong High Plains was first reported in 2004 (Moore *et al.* 2017). As well as being intensive (many seedlings in a given location), seedling germination was widespread. In an unpublished May 2005 report describing willow control work undertaken that summer (Mandar Services report to NECMA on works), one contractor noted that 'There was no sector of the high plains that did not have seedlings present if there was a suitable germination environment.' Subsequent assessments of population structure suggested that seedling germination in burnt peatlands was high for at least four years after the 2003 fires (Moore and McAllister 2013). The response from land management agencies was swift, with a control program established in the summer of 2004/2005 that continues to this day.

Managing *Salix cinerea* invasion in alpine peatlands

Control work undertaken in the first two years of the control program focused on removing mature individuals at the headwaters of waterways (e.g. Rocky Valley Creek, Pretty Valley Creek, West Kiewa River and Big River) close to the Bogong High Plains as well as seedling removal within peatlands, especially in the Rock Knobs area. Much of the seedling removal was undertaken by volunteers. However, the extent and intensity of germination after the 2003 fires meant that it was not possible for land management agencies to treat the invasion in all locations simultaneously. For example, with respect to willow control on the Bogong High Plains,

it became clear that Parks Victoria would need to transition from a relatively ad hoc control strategy to a more measured plan for control over the subsequent 5–10 years (Moore *et al.* 2017). In collaboration with researchers and students at the United States Geological Survey, the University of Melbourne and Monash University, I have addressed two key questions. The first question was how to divide effort between managing mature populations that could facilitate further invasion and newly emerging seedlings on the Bogong High Plains. The second question was where to prioritise removal of seedlings from the Bogong High Plains, given that it was not possible to search the entire area in one year. How each of these questions was resolved is described in the following sections.

Source vs seedlings?

In the hope of minimising further invasion, a large proportion of the initial resources were focused on controlling mature willow in the river reaches, with less effort allocated to managing seedlings on the Bogong High Plains. As the full scope of seedling establishment was realised, the focus shifted. Seedlings were found in pretty much any peatland or wet location that had been burnt. There was an imperative to focus on removing as many of the seedlings as possible before they could make an impact—before they could mature and produce seed themselves—and while they were relatively easy to control.

It was not easy to identify how to allocate effort between the source (seed producing) populations and the seedling populations to minimise the impact of the invasion overall. In part this was because the ideal balance depended on a range of factors, including the dispersal ability of the seed, the location of existing mature populations, the rate at which seedlings grew and matured on the Bogong High Plains, and the effectiveness of control, all of which were poorly understood (Moore and Runge 2012). Furthermore, the ideal balance also depended on what the overall goal of management was. For example, was the aim to eradicate the species from the region or minimise damage to biodiversity?

This problem was addressed by using structured decision-making to examine the problem and identify a solution that best met the goals of managers (Moore and Runge 2012). Structured decision-making is a participatory process designed for making management decisions. It simplifies the decision-making process by breaking down the decision at hand into a series of steps that are worked through sequentially (Keeney 1992; Gregory *et al.* 2012). A key feature of structured decision-making is that it clearly separates the objective of management (a values-based judgement with no single answer) from the measurable assessment of the effectiveness and feasibility of different management alternatives (a fact-based activity with a specific, if uncertain, answer). A structured decision-making workshop was held in October 2009 to develop a willow control strategy. The workshop included staff from all organisations engaged in willow control on and around the Bogong High Plains at the time, specifically Parks Victoria as well as the Department of Environment and Primary Industries (DEPI, now Department of Environment, Land, Water and Planning [DELWP]), NECMA and Falls Creek Resort Management (Moore *et al.* 2017).

A key outcome of the structured decision-making process was that it clarified the objective (Moore *et al.* 2017). It was not considered feasible to undertake eradication. Rather, the key driver was the protection of the sub-alpine and alpine peatlands that were already threatened by willow invasion and had high conservation value. This then enabled Parks Victoria to zero in on a control strategy that minimised the expected amount of willow in peatlands over the long term. Moore and Runge (2012) built a computer model of the system to predict the anticipated amount of willow invasion in peatlands over 200 years for a range of different willow control strategies. This model accounted for current willow distribution, fire frequency (since fire facilitates seedling establishment), seed dispersal distance, willow growth rate and control effectiveness. The model also accounted for our uncertainty regarding these system attributes. It was possible to test the benefit of allocating effort to four control regions: 1. peatlands; 2. other suitable habitat on the Bogong High Plains; 3. river reaches adjacent to the

Bogong High Plains that were more expensive to manage because they are in steep and inaccessible country; and 4. all remaining river reaches within a 10 km radius. The analysis revealed that for the current levels of resources, the optimal strategy was to allocate all effort to control willow seedlings in the peatlands. This reflected the fact that resources were considered insufficient to manage enough of the source populations to avoid re-invasion of peatlands. It also reflected the fact that peatlands already had large seedling populations which needed to be controlled if impacts were to be avoided (Moore and Runge 2012). These findings shifted where control work was focused and by the summer of 2013/2014 all Parks Victoria-funded control work was being undertaken within peatlands on the Bogong High Plains.

The study included two different kinds of analysis to examine how uncertainty regarding how the system worked affected the model predictions and the subsequent optimal strategy. Moore and Runge (2012) found that for up to three times current budget levels, the optimal management strategy stayed the same. Only if very large budgets were available did the optimal strategy change to include control in other areas. For these large budgets, value of information analysis revealed that it was the dispersal distance of willow seeds followed by the recovery rate of post-fire vegetation cover that had the largest effect on the optimal strategy. This uncertainty analysis was important as it meant Parks Victoria could be confident that, given available resources, the strategy was likely to provide the best possible outcome despite a lot of uncertainty (Moore *et al.* 2017).

Spatial allocation of search effort

The second issue was to identify where to look for seedlings on the Bogong High Plains as it was clear that the entire area could not be searched at once. The question then was: where and how long to look first? 'Where' was not so tricky. Intuitively, it was to look in places where willows were most likely to be and where controlling willows would have the greatest benefit (peatlands), but 'how long' to look was a more difficult question. Answering this came down to the idea that you will find more of something (e.g. willow seedlings) the longer you look

because detection is imperfect (Moore 2015). Experiments have shown that, when searching for plants, the probability of finding all individuals increased with time (Garrard *et al.* 2008; Moore *et al.* 2011), and monitoring of willow control showed that detection of willow seedlings wasn't perfect either (Giljohann *et al.* 2009).

My collaborators and I (Giljohann *et al.* 2011) applied a decision framework developed by Hauser and McCarthy (2009) to calculate where and for how long to look for willows on the Bogong High Plains. We built a habitat suitability model to predict the likelihood of willow occurrence across the Bogong High Plains that accounted for fire history and previous control effort. We accounted for imperfect detection using field studies to estimate the proportion of willows controlled per unit effort in the field (Giljohann *et al.* 2011). To reflect the objective of removing seedlings from peatlands first, we weighted controlling willows in peatlands twice the benefit of other locations. Using these three inputs, we made spatial recommendations regarding where and for how long to look over the summer of 2009/2010 (Giljohann *et al.* 2009) and updated these twice to account for control works undertaken in the summer of 2010/2011 and 2011/2012 (Fig. 2). One of the major insights from this work was that contractors were tending to spend too long at each location trying to find every last small seedling. This was inefficient as it was known that all sites would need to be visited again in subsequent years. To address this, Parks Victoria directed contractors to search for and control willows if they were taller than knee height (although incidental control of smaller willows also was acceptable). This freed contractors from the expectation that they must find every single individual and allowed them to cover more area.

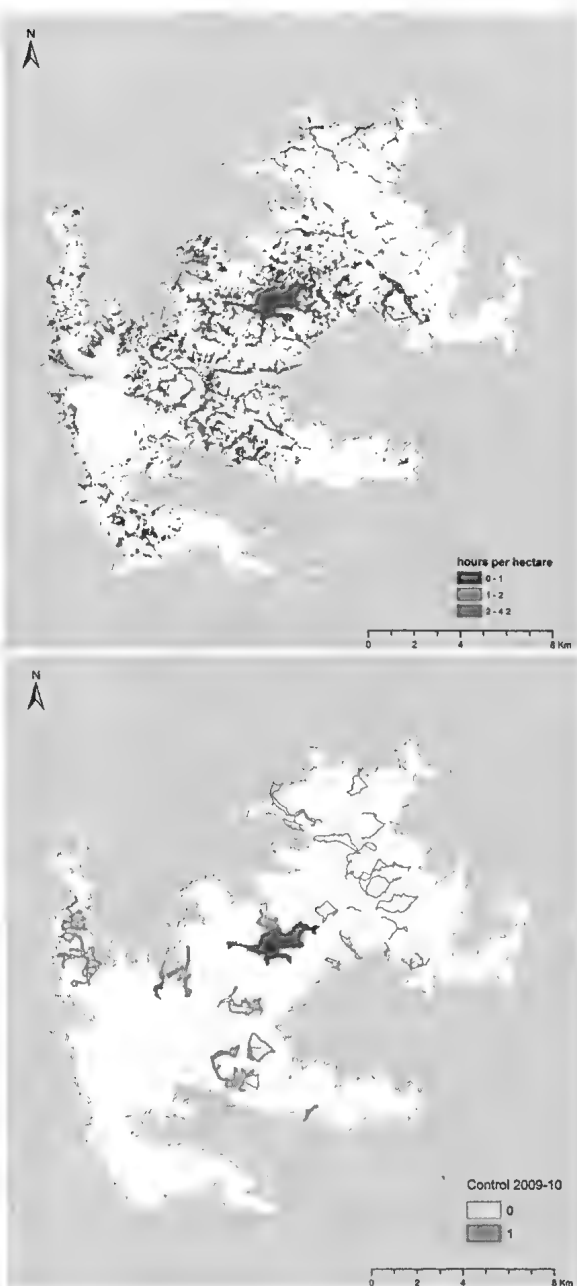


Fig. 2. Top: the Bogong High Plains study area showing the recommended spatially prioritised control effort for the 2009/2010 season assuming an annual budget of 400 person days, with control in peatlands weighted twice that of other regions. Bottom: the subsequent control effort undertaken in the 2009/2010 season based on GPS tracks of contractors. After Giljohann and Moore (2011).

The increased focus on seedling removal from 2012/2013, along with monitoring that showed declining seedling densities, meant that it was now feasible to implement a search strategy to revisit peatlands at least once every five years. It was estimated that it would take at least five years for individual plants to mature and start producing seed because growth rates of seedlings monitored in peatlands were relatively slow (Moore and McAllister 2013). The rotation approach was implemented by splitting the Bogong High Plains into five sections and searching all peatlands in one section per year on a five-year rotation (Parks Victoria 2014).

Status and future prospects

By the end of the 2020/2021 season, all peatlands will have been searched at least twice. Willow seedling densities in peatlands now are low and vegetation cover of peatlands burnt in 2003 and 2006 has largely recovered. So, where to next?

There is still a substantial amount of seed-producing willow in the region, meaning that ongoing monitoring and control is necessary. Contractors have reported that there are still many willow infestations in the Bogong High Plains reaches (Parks Victoria, pers. comm. 15 April 2020). Hence, Parks Victoria is now looking to expand control effort to include all willows on the Bogong High Plains, not just those in peatlands. To assist with targeting likely locations, my research team is currently updating the spatial prioritisation from 2009–2010. The updated prioritisation will take account of the extensive effort allocated to peatlands over the last decade as well as the change in focus to manage willow across all suitable habitat on the Bogong High Plains. Furthermore, some peatlands on the Bogong High Plains were burnt in 2019. These peatlands should be checked for seedlings during the next couple of summers.

Catchment Management Authorities, Parks Victoria and DELWP allocate substantial resources to manage extensive willow infestations in eastern Victoria. However, containment and control is an ongoing issue as willows spread easily and often reinfest managed areas. Of particular concern are willow infestations in the headwaters of streams and rivers that act as a source for new infestations downstream, but

that can also provide a seed source for spread across catchments or into alpine and sub-alpine vegetation (Hopley and Young 2015). Research is underway to provide improved estimates of dispersal distance and, importantly, to understand how regional topography might influence the likely direction and distance of seed dispersal. This information is important for developing a landscape-scale, cross-jurisdictional strategy for willow monitoring and control.

As this paper makes clear, one of the big challenges of *Salix cinerea* management is that these willows produce seeds. Naturally, much of the management effort and research has focused on the two species that produce seeds in Australia (Parks Victoria 2014). However, hybrids are likely to be widespread (Cremer 2003), but have yet to be systematically studied, and their ability to produce seed is largely unassessed. Further assessment of the distribution and risk posed by wild hybrids is an important activity for future research.

Conclusion

Willows have been widely planted in riparian systems to stabilise soils and reduce erosion, but have become an environmental weed, spreading into natural riparian vegetation. Furthermore, the large alpine fires of 2003 triggered a widespread invasion of one willow species, *Salix cinerea*, into threatened alpine and subalpine peatlands. Extensive control of seedlings has occurred over the past 15 years as well as control of seed source populations. This program has utilised research undertaken to set management priorities and make the most of management resources. Despite substantial effort and much progress, willows are still widespread in eastern Victoria and their effective management remains a priority for land management agencies.

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One hundred and six years ago

EXCURSION TO BAW BAW

At Coalville signs of a former industry of the place indicated by the name were seen. Just beyond Narracan is a fine fall on the creek of about 25 to 30 feet, over which a good volume of water was pouring. Here a peculiar growth of bright pink, grass-like stems was covering some of the basalt stones near the water's edge, which puzzled me; but on showing it to Mr. French later, he immediately asked if any willows grew near; I said "Yes," so my pink grass turned out to be willow roots.

From *The Victorian Naturalist* XXX, p 207, March 5, 1914

A method for safely removing the invasive kelp *Undaria pinnatifida* (Harvey) Suringar

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Abstract

The invasive kelp *Undaria pinnatifida* (Harvey) Suringar can be difficult to eradicate due to a short period of rapid growth prior to reproductive maturity and a high output and longevity of zoospores (motile spores). A method to safely isolate the reproductive sporophyll (spore-bearing structure) was proposed to reduce spread of zoospores during management efforts, and an experiment undertaken to verify that sporophylls would not develop from remaining tissue left in situ. All tissue grew while gradually senescing and being grazed until no tissue remained. No regrowth of the sporophyll occurred. This method offers a more biosecure way of removing *Undaria pinnatifida* individuals in an active management program. (*The Victorian Naturalist* 137(6), 2020, 246–251)

Keywords: active management, marine macroalgae, temperate subtidal reef, *Undaria pinnatifida*, invasive kelp

Introduction

The Japanese Kelp or Wakame *Undaria pinnatifida* (Harvey) Suringar (hereafter *Undaria*) is an internationally recognised invasive marine species (South *et al.* 2017), and an Australian priority marine pest (Marine Pest Sectoral Committee 2018). The sporophyte (spore producing generation or 'plant') consists of a holdfast (attachment organ), stipe (stem between holdfast and blade) and blade (Fig. 1), and can grow up to three metres in length, but is often smaller. Sporophylls (structures producing spores) occur laterally along the stipe (Fig. 1), usually along the lower section, but can extend the full length (Alvarez and Boraso 2020). Native to Japan, Korea, China and Russia, *Undaria* is a prized food product, with extensive aquaculture meeting production needs (Kain 1991; Epstein and Smale 2017). Thought to have been spread through ballast vectors, it has become established in thirteen countries, including Australia, with impacts varying from benign to negative (Epstein and Smale 2017). *Undaria* establishes in areas where native algae are absent or disturbed, and favours high nutrient, sheltered environments (Carnell and Keough 2014; Crockett *et al.* 2017).

Eradication attempts mostly have been unsuccessful, due to high reproductive output (Schaffelke *et al.* 2005; Hunt *et al.* 2009; Primo *et al.* 2010), longevity of zoospores

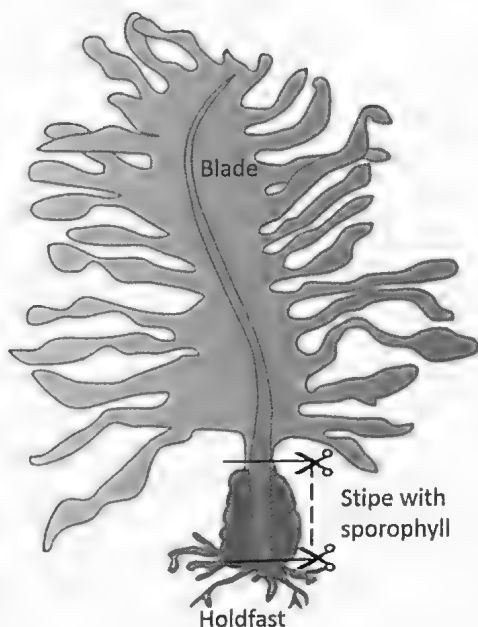


Fig. 1. Diagram of *Undaria pinnatifida* individual showing sporophyte structure and excision points of stipe. Modified from Tracey Saxby, Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/imagelibrary/).

(motile spores) and logistically challenging and expensive removal options that hamper efforts (Department of Water Resources [DAWR] 2015; Crockett *et al.* 2017). The majority of reported removal attempts use hand pulling of sporophytes and retention in mesh catch bags for land-based disposal (DAWR 2015; Crockett *et al.* 2017; South *et al.* 2017). These methods are not able to remove microscopic gametophytes, which produce egg and sperm that result in another sporophyte generation following fertilisation. Also, if sporophytes are reproductive, removal using a catch bag may aid spread of the species, by loss of zoospores through the mesh of the catch bags—containing removed sporophytes—carried by the divers as they swim). On land, weeds are usually treated before flowering. In *Undaria*, there can be a short onset to reproductive maturity from the time juveniles can first be recognised in the growth season and monitoring relies on SCUBA or snorkelling. Given these logistical difficulties in identifying sporophytes within the reproductive time-frame, a method to remove them while reproductive is needed.

Could a method akin to terrestrial weed management, e.g. 'dead-heading', reduce the chance of zoospore spread at treatment sites? If so, this would provide a more biosecure method of *Undaria* removal. Parsons (1995) found it was unclear whether *Undaria* could regrow from a holdfast or cut stipe. Gao *et al.* (2013) identified that removal of the upper portion of the blade led to increased growth, as long as 30 cm of the thallus remained above the sporophylls. This was beneficial in aquaculture but would be detrimental in removal attempts. In the laboratory, whole reproductive plants were regrown from callus tissue sourced from the meristem (Kawashima and Tokuda 2004). This project, therefore, examined whether excision and removal of only the stipe, along with any sporophylls, would be a more viable method of management than removal of the whole sporophyte by determining whether any sporophylls would develop from holdfasts or blades left in situ.

Methods

Samples were collected from reef at a depth of 10–11 m at Pope's Eye (Port Phillip Heads

Marine National Park) in Victoria (38° 16' 37.20" S, 144° 41' 56.40" E) using SCUBA as part of an ongoing management program. Three whole individuals were carefully removed from the substrate (including the holdfast), retained in a sealed ziplock bag and returned to shore.

Each individual had the stipe, including any sporophylls present, excised with a cut at the top and bottom of the stipe (Fig 1). Any part of the stipe without sporophylls was retained and placed in a mesh nylon bag (mesh approximately 2–3 mm) along with the holdfast and blade and attached to the underside of a pontoon at Queenscliff harbour (in a similar position to other *Undaria* that had colonised there) and periodically examined (as convenient) for any regrowth or development of sporophylls.

Results

Sporophylls did not develop on any tissue, and no tissue remained by day 132. The blade grew for at least 32 days but signs of senescence were apparent on one blade at day 14 and on all three blades at day 32 (Table 1, Fig 2). Grazing damage was apparent at day seven. The stipe and holdfast of two sporophytes no longer remained by day 32. Growth of each organ occurred over the first 32 days, while specimens remained. Grazing was visible at day seven for one specimen and day 14 for the other, with senescence visible only at day 32 for the last remaining stipe and holdfast. Crustacean grazers observed included the Sea Centipede *Euidotea peronii*, amphipods and isopods.

Discussion

Marine invasive species continue to be difficult to eradicate or control, in part due to inadequate surveillance, difficulty in identification, or inability to respond rapidly before they become too abundant or further expand their distribution. The added logistical difficulties of accessing marine habitats and what responses are possible, effective and targeted while in water also pose barriers to successful management. Reducing zoospore spread should be part of any effective management response. *Undaria* sporophytes can release zoospores when stipe width is only 1 cm (Schaffelke *et al.* 2005; Primo *et al.* 2010), the second smallest size class of five defined by Schaffelke *et al.* (2005).

Table 1. Growth, grazing and senescence of *Undaria pinnatifida* following excision of the stipe. The holdfast, blade and any stipe tissue without sporophylls was attached beneath a pontoon at Queenscliff, Victoria for 132 days. + present, - absent.

	Plant (no.)	Day 7 31 Aug 2018			Day 14 7 Sept 2018			Day 32 25 Sept 2018			Day 132 3 Jan 2019	
		Holdfast	Stipe	Blade	Holdfast	Stipe	Blade	Holdfast	Stipe	Blade		
Growth	1	none remaining	none remaining	+	none remaining	none remaining	+	none remaining	none remaining	+		
	2	+	+	+	+	+	+	none remaining	none remaining	+		
	3	+	+	+	+	+	+	+	+	+		
Grazing damage	1	none remaining	none remaining	+	none remaining	none remaining	+	none remaining	none remaining	+		
	2	-	-	+	+	+	+	none remaining	none remaining	+		
	3	+	+	+	+	+	+	+	+	+		
Senescence	1	none remaining	none remaining	-	none remaining	none remaining	-	none remaining	none remaining	+		
	2	-	-	-	-	-	-	none remaining	none remaining	+		
	3	-	-	-	-	-	+	+	+	+		
Crustacean/ herbivore presence	1		+			-			-			
	2		-			+			+			
	3		-			-			+			
Sporophyll development	1-3											

No tissue remaining

None throughout

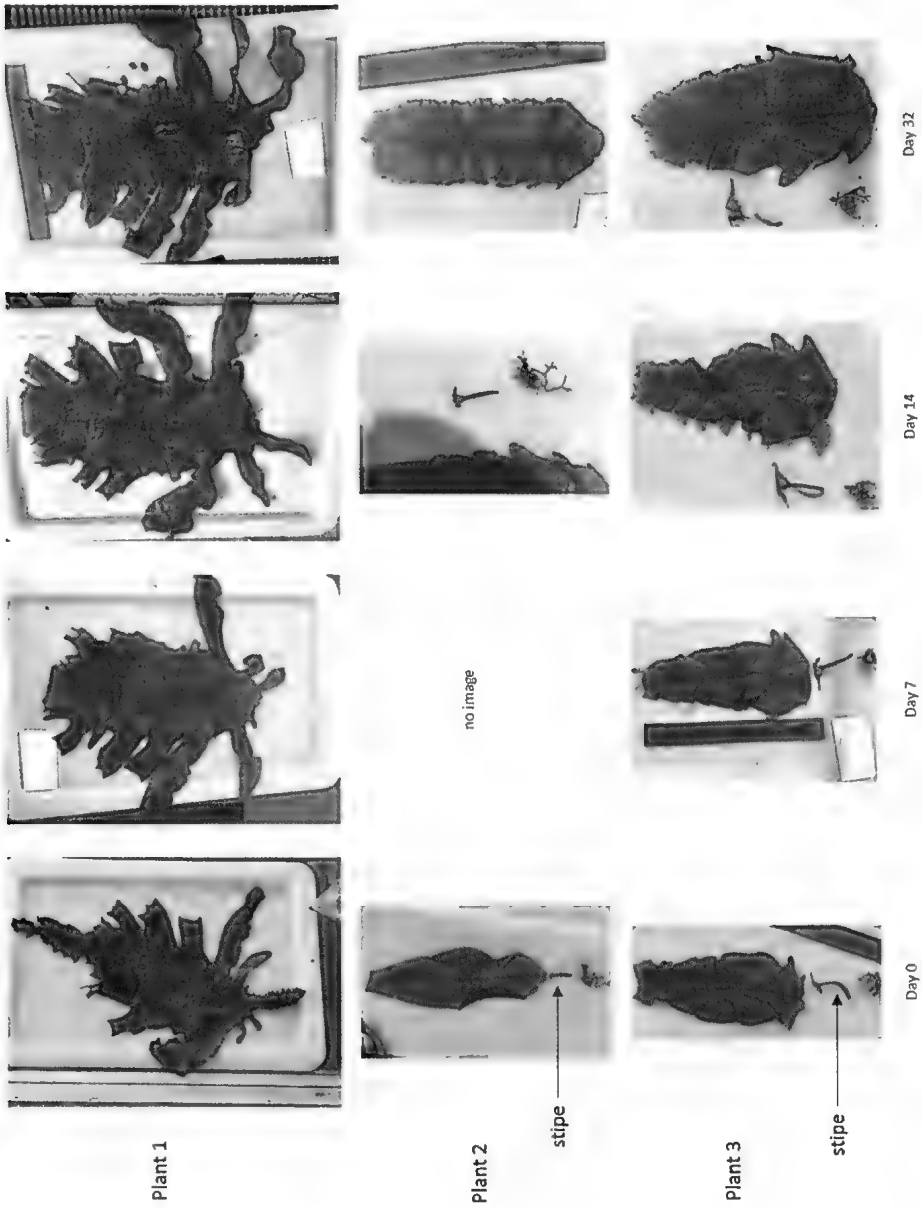


Fig. 2. Images of plants from day 0 to day 32 showing herbivory, senescence and lack of sporophyll development.

In a Port Phillip Bay study (Primo *et al.* 2010) all size classes had a similar spore competency per month, ranging from 0.13×10^5 spores $\text{cm}^{-2} \text{h}^{-1}$ to 7.33×10^5 spores $\text{cm}^{-2} \text{h}^{-1}$.

There is much debate as to whether *Undaria* has a detrimental ecological impact on native marine life when in new locations. Epstein and Smale (2017) reviewed seventeen papers investigating impacts and found a range of reported conclusions, the majority of which agreed that *Undaria* was most common in disturbed environments, and excluded where native algae dominated. The variety of conclusions are complicated also by a large range in experiment duration, ranging from 1 to 48 months.

Indirect negative impacts of *Undaria* have been examined only recently. In Port Phillip Bay, Victoria, Reeves *et al.* (2018) found that *Undaria* was inferior to the native *Ecklonia radiata* (hereafter *Ecklonia*) when considering sweeping of substrate, which prevents the establishment of algal turf (Reeves *et al.* 2018). Algal turf growth can initiate an exclusionary feedback loop preventing *Ecklonia* return (Reeves *et al.* 2018). This could be due to *Undaria* having a thinner, more 'papery' frond in comparison to the tougher *E. radiata*. *Undaria* is also an annual species while *Ecklonia* is perennial, therefore providing a year-long sweeping service as well as persistent habitat for associated biota. In addition to providing habitat for a much shorter period of time, *Undaria* holdfasts host different assemblages of biota in comparison to the native *Ecklonia* (Howland 2012), indicating further indirect impacts. The culmination of indirect impacts could lead to a larger than assumed impact on native marine communities. Until these indirect impacts are further understood and long-term, well-designed and repeated experiments with a substantial sample size can demonstrate that impacts are minimal, it is preferable to prevent further spread and continue active management in newly infested locations or sites of high environmental value.

The results of this experiment, albeit with only a small sample size, support sporophyll excision and retention as an improved technique for *Undaria* removal. In addition to preventing zoospore spread, it would reduce manual handling, allow tissue to be consumed or naturally

break down, and reduce disturbance (as the holdfast can remain in place), thereby minimising the likelihood of new *Undaria* establishment. This method has been adopted within an adaptive management program underway at Pope's Eye within Port Phillip Heads Marine National Park in Victoria, Australia. Further trials at additional locations are recommended to further assess outcomes.

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Bandicoots versus Kangaroos: those who cannot remember the past are condemned to repeat it

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Abstract

The Back Paddock of Woodlands Historic Park has formed a critical component of the recovery program for endangered Eastern Barred Bandicoots *Perameles gunnii* (Bandicoots). Two attempts have been made to reintroduce Bandicoots to this site. The first was in the 1990s: over 100 Bandicoots were released and a robust population became established. However, intense grazing pressure by Eastern Grey Kangaroos *Macropus giganteus* (Kangaroos) and European Rabbits *Oryctolagus cuniculus*, exacerbated by dry conditions, degraded the grassy woodlands that provide secure nesting and foraging habitat for Bandicoots. Kangaroos were eventually culled, but this intervention could not prevent a precipitous decline of Bandicoots to undetectable levels. Another attempt to reintroduce Bandicoots occurred 20 years later. Rabbit control was far more effective this time and a new Bandicoot population became established. However, the Kangaroo population also increased rapidly, grass cover became severely reduced, and the Bandicoot population collapsed for the second time. All Kangaroos were eventually removed by the end of 2019, despite protests by animal activists, and the Bandicoot population is now recovering. (*The Victorian Naturalist* 137(6), 2020, 251–257)

Keywords: Eastern Barred Bandicoot, Eastern Grey Kangaroo, endangered species, habitat degradation, population control.

Introduction

‘Those who cannot remember the past are condemned to repeat it’ (Santayana 1905, p. 284). Just two years after the American philosopher and poet George Santayana composed that famous aphorism, *The Victorian Naturalist* published a paper by Isaac Batey about life on the other side of the world. Batey (1907) described changes in native mammal populations from the time he arrived in the Sunbury district, Victoria, in 1846. He wrote that the ‘Short-tailed Bandicoot’, now known as the Eastern Barred Bandicoot *Perameles gunnii*, was ‘not plentiful’ at first and ‘became rarer still’, then ‘again became numerous’ 20 years later (p. 71).

He also noted that ‘Kangaroos and Wallabies’, i.e. Eastern Grey Kangaroo *Macropus giganteus* and Black Wallaby *Wallabia bicolor*, ‘were never seen in the area’ from 1846, probably because they were ‘easily frightened or driven away by the first settlers’ in the open landscape (Batey 1907, p. 71).

This paper presents a modern history of Eastern Barred Bandicoots and Eastern Grey Kangaroos (hereafter Bandicoots and Kangaroos) at Woodlands Historic Park, which is situated no more than 10 km from the core of the district described by Batey (1907). We outline two programs that have attempted to

reintroduce Bandicoots to this landscape. We also describe the ecological impacts of Kangaroos, and the threats they have posed to the Bandicoot reintroduction program.

Bandicoots and Kangaroos

The Eastern Barred Bandicoot (hereafter Bandicoot) is native to the basalt plains of south-west Victoria, but is classified as extinct in the wild on mainland Australia, primarily due to habitat loss and predation by the introduced Red Fox *Vulpes vulpes* (hereafter Fox). Bandicoots weigh an average of 750 g and, therefore, fall into the Critical Weight Range for Australian mammals, making them particularly prone to extinction (Burbidge and McKenzie 1989). Bandicoots are solitary, nocturnal and opportunistic omnivores, eating mostly invertebrates, even crabs, but also some plant matter, including roots, bulbs and fruit (Dufty 1994; Loeffler 2018). Preferring to dig for their food, they are regarded as ecological engineers because their diggings improve soil health (Halstead *et al.* 2020). Their habitat requirements are flexible, needing only open areas for foraging adjacent to structurally complex areas for nesting (Winward *et al.* 2013). Bandicoots are capable of populating areas rapidly due to their 12.5-day gestation, 2–3 young born per litter and continuous breeding, allowing up to five litters to be produced per year; these young can also breed from three months of age (Backhouse *et al.* 1994). These habitat, diet and reproductive characteristics make this species easy to reintroduce, provided that release sites remain fox-free (Coetsee 2016). Populations are currently found in four reserves within the indigenous range, surrounded by predator exclusion fencing (Hamilton Community Parklands, Mount Rothwell, Tiverton and Woodlands Historic Park) and three islands outside their original range (Churchill, Phillip and French Islands).

The Eastern Grey Kangaroo (hereafter Kangaroo) is widespread in Eastern Australia, extending from Cape York to Tasmania. Classified as Least Concern on the IUCN Red List, this species is common through much of its range, and is considered overabundant in some situations (Eldridge and Coulson 2015). It is well represented in reserves and, although some habitat has been lost to urbanisation and intensive ag-

riculture, this species has benefited from clearing of forest, pasture improvement, provision of water and suppression of predators (Eldridge and Coulson 2015). Ideal habitat is a mix of cover and open foraging areas, including forest, woodland, mallee, shrubland, pasture, crops, plantations and golf courses. Kangaroos can weigh up to 90 kg (Eldridge and Coulson 2015). They are crepuscular and gregarious, forming large groups at high densities, feeding mainly on grasses, as well as forbs, shrubs and trees (Eldridge and Coulson 2015). Females mature from 18 months (Poole and Catling 1974) and breed seasonally, giving birth to a single young that leaves the pouch after 10 months and is weaned after 18 months (Poole 1975). Kangaroos can live for at least 12 years, but many do not reach old age (Quin 1989). Although their reproductive rate is much lower than that of the Bandicoot, a population of Kangaroos can double in as little as 1.5 years in optimal habitat without predators (Coulson 2001).

Woodlands Historic Park

Gellibrand Hill Park, situated immediately north of Melbourne Airport, was reserved in 1981 under the *National Parks Act 1975*. Additional areas were reserved in 1986 and 1995, and the park was renamed Woodlands Historic Park, in reference to the heritage-listed Woodlands homestead and outbuildings, built in 1843. Woodlands Historic Park is now 820 ha in area. The park is open to the public at all hours and is managed for a wide range of uses, including passive recreation and conservation of natural and heritage values.

The park lies on the boundary of the Central Victorian Uplands and Victorian Volcanic Plains bioregions, and supports a number of vegetation associations: Red Gum *Eucalyptus camaldulensis* woodland; Red Gum riparian and wetland complex; Grey Box *E. microcarpa* woodland; mixed-species woodland of Red Gum, Grey Box, Yellow Gum *E. melliodora* and Manna Gum *E. viminalis*; and mixed-species shrubland of Drooping Sheoak *Allocasuarina verticillata*, Hedge Wattle *Acacia paradoxa* and Sweet Bursaria *Bursaria spinosa*. There are also sizeable patches of perennial grassland dominated by Kangaroo Grass *Themeda triandra* and Weeping Grass *Microlaena stipoides*, to-

gether with a variety of exotic grasses. These vegetation communities provide ideal habitat for both Bandicoots and Kangaroos.

Woodlands Historic Park was chosen as the first site for reintroduction of Bandicoots. It was considered an ideal location, as it was managed by the then Department of Conservation and Environment, with full-time staff and on-site facilities (Seebeck 1990). In 1987, a 1.8 m high predator-barrier fence was constructed around 400 ha of open grassy woodland, which became known as the Back Paddock, representing the less developed portion of the property (Seebeck 1990). This area was later reduced to 300 ha to exclude a weak point where the fence crossed Greenvale Creek. Aside from its intended role of excluding predators, this fence also enclosed a population of about 30 Kangaroos, six deliberately introduced for a study of the interactions between grazing and burning, and the others of local origin (Coulson 2001).

From 1988 to 1991, Woodlands was used as the primary captive breeding facility for Bandicoots. This facility comprised a set of 16 predator-proof pens constructed inside the Back Paddock to receive Bandicoots extracted from the last remaining wild population at Hamil-

ton. Although breeding occurred, the facility was abandoned in 1991 after a number of injuries to Bandicoots caused by the fence, a Black Rat *Rattus rattus* invasion due to supplementary feeding, and difficulties monitoring breeding and health of the Bandicoots (Winnard and Coulson 2008). The captive breeding program was then relocated to properties managed by the Zoological Board of Victoria.

The first reintroduction

In the Back Paddock, intensive predator control was carried out by shooting, poisoning and trapping in preparation for the release of Bandicoots. In total, 104 Bandicoots were released into the Back Paddock from 1989 to 1992 (Winnard and Coulson 2008). Bandicoot abundance, measured as trap success, reached its peak in 1996 (Fig. 1), when over 600 were estimated occupying the entire reserve. The success of the reintroduction program was attributed to the control of introduced predators and European Rabbits *Oryctolagus cuniculus* (hereafter Rabbits), combined with a run of good seasons with enhanced soil moisture and invertebrate resources (Winnard and Coulson 2008).

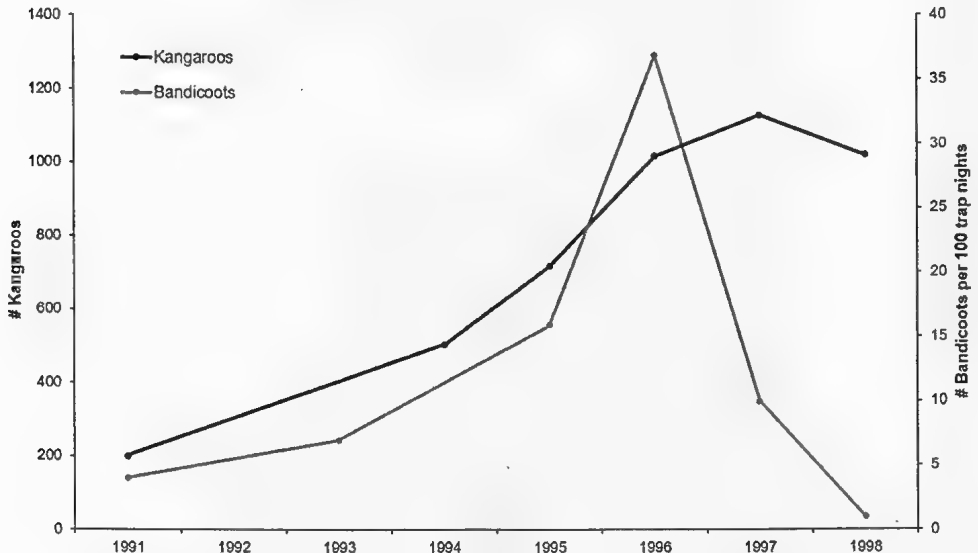


Fig. 1. Abundance estimates for Eastern Barred Bandicoots (EBB) *Perameles gunnii* and Eastern Grey Kangaroos *Macropus giganteus* in the Back Paddock, Woodlands Historic Park, from 1991 to 1998.

However, the Bandicoot population then underwent a precipitous decline (Fig. 1). Predation was not considered a major factor contributing to this alarming trend. Rather, the grass cover that Bandicoots needed to nest and forage safely was degraded by the rapidly increasing populations of Kangaroos (Fig. 1) and Rabbits. This process was apparently exacerbated by dry conditions and removal of at least 100 Bandicoots between 1995 and 1997, to supplement other sites, which may have hastened the decline (Todd *et al.* 2002). By 1998, only two Bandicoots could be trapped (Fig 1); these individuals were taken into captivity. Some Bandicoot diggings were subsequently observed, so 18 Bandicoots in total were released into the Back Paddock in 1999 and 2001 (Long *et al.* 2005). Numbers remained low: by 2006, no Bandicoots could be trapped; only one was seen by spotlight; and diggings were rare (Winward 2010). The population was considered to be functionally extinct.

The second reintroduction

In 2012, the predator exclusion fence was upgraded and realigned to exclude Gellibrand Hill, a sandy area prone to Fox incursions, further reducing the Back Paddock to 235 ha. An

internal Rabbit-proof fence was also installed to support intensive Rabbit control efforts in the western portion of the Back Paddock. In 2013, once the reserve had been Fox-free for three months, 42 Bandicoots were released. Bandicoots have since been monitored by trapping on a grid covering the Back Paddock at least twice per year, but the efficacy of this method has been reduced by a high, non-target capture rate of Common Brushtail Possums *Trichosurus vulpecula*. Distance sampling by spotlight on parallel transects across the area has, therefore, been used as a complementary survey method.

Distance sampling indicated that, by 2016, the Bandicoot population had risen to an estimated size of 502 (± 207). This coincided with the most successful trapping survey, with 142 Bandicoots caught over four days (Fig. 2). The population then steadily declined. In February 2018, there were insufficient sightings on the transect lines to estimate density by distance sampling despite high detectability of Bandicoots in the sparse grass cover. By March 2019, only eight individuals were caught over four days (Fig. 2). This decline coincided with lower-than-average rainfall and a dramatic decline in grass cover across the Back Paddock (Fig 3). Some Fox in-

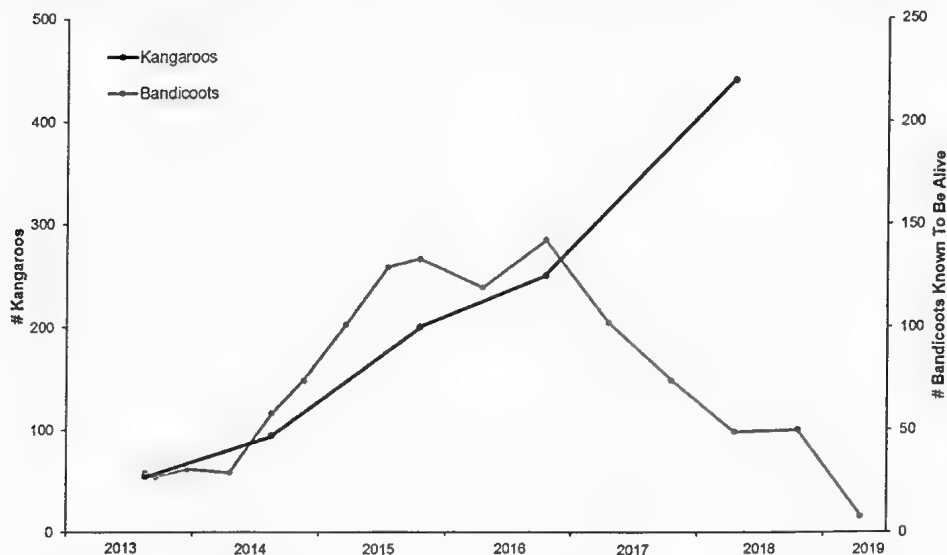


Fig. 2. Abundance estimates for Eastern Barred Bandicoots (EBB) *Perameles gunnii* and Eastern Grey Kangaroos *Macropus giganteus* in the Back Paddock, Woodlands Historic Park, from 2013 to 2019.



Fig. 3. Grass cover at the same photo point on the Bandicoot trapping grid in the intensive Rabbit control area of the Back Paddock, Woodlands Historic Park. Top: February 2014; Middle: April 2018; Bottom: December 2020. Photos Travis Scicchitano.

cursions occurred during this period, but these were immediately detected and Foxes were removed swiftly, so were not thought to have had any real impact on the Bandicoot population. The more likely driver of the Bandicoot decline was the loss of essential grassy habitat, reducing foraging cover, nest sites and food availability.

Parks Victoria recognised that heavy grazing by Kangaroos was a threatening process for Bandicoots, as well as other components of biodiversity in the Back Paddock. In the lead-up to the second Bandicoot release, Parks Victoria had conducted a program of regular surveys and intermittent culling with the aim of holding the population below 50 Kangaroos. Although the Kangaroo population was at that target number when the Bandicoots were released, it increased rapidly over the following four years (Fig. 2). By 2018, the Kangaroo population had reached 445 (Fig. 2) and a dramatic loss of grass cover was apparent (Fig. 3). Conscious of Santayana's (1905) aphorism, Parks Victoria was determined not to allow history to repeat itself. Non-lethal control options for Kangaroos were explored, but fertility control could not produce the immediate reduction in Kangaroo abundance that was needed to restore Bandicoot habitat. Efforts to cull Kangaroos were thwarted a number of times by protests led by the Australian Society for Kangaroos, while habitat condition deteriorated further and the Bandicoot population crashed.

Parks Victoria conducted covert culling operations from late 2018, and all Kangaroos had been removed from the Back Paddock by late 2019. Habitat recovery in 2020 has been dramatic following significant autumn rainfall (Fig. 3). The fate of the Bandicoot population is less clear, because surveys were postponed due to the COVID-19 pandemic. However, foraging digs are evident throughout the Back Paddock and Bandicoots have been detected by spotlight and camera trapping, suggesting that the population has persisted.

Conclusion

Of the two attempts to reintroduce Bandicoots to the Back Paddock, the first ended in failure but the second seems likely to succeed after intervention at a critical stage. In both cases, inadequate or belated control of Kangaroos

allowed their populations to increase to very high densities, resulting in habitat degradation that undoubtedly contributed to Bandicoot declines. Such situations, where the viability of a threatened native species is compromised by another native species, are becoming increasingly common (Woinarski 2019). This management challenge is amplified where management actions may be challenged by animal rights groups, which do not place the same value on species conservation and ecosystem function. We argue that the benefits for a rare, restricted species like the Eastern Barred Bandicoot must outweigh the costs to a common and widespread species like the Eastern Grey Kangaroo.

Acknowledgements

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Releasing an Eastern Barred Bandicoot *Perameles gunnii*. Photo Zoos Victoria

An introduced kelp has less impact on a native algal community than overgrazing by native sea urchins

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Abstract

The introduced kelp Wakame *Undaria pinnatifida* is found throughout Port Phillip Bay, where it grows on a variety of hard substrates including natural reef, cobble and shell, as well as pylons, floating pontoons and boat hulls. We studied the efficacy of *Undaria pinnatifida* (hereafter *Undaria*) removal strategies in Jawbone Marine Sanctuary, Williamstown, Victoria, from June 2013 to July 2014, inclusive. The study examined the frequency and timing of removal of *Undaria* on the native algal canopy in Jawbone Marine Sanctuary. Recruitment and growth of *Undaria* was unexpectedly low in 2013. *Undaria* recruited in higher abundances to areas with a higher amount of bare space and lower cover of native algae. All removal treatments effectively reduced the number of thalli in plots, but this had no effect on the per cent cover of *Undaria* in subsequent surveys. The abundance of *Undaria* at the end of the experiment (July 2014) was lower than at the start (June 2013) in all treatments and the control. The greatest influence on the subtidal macroalgal community during the study was grazing by the Shortspined Sea Urchin *Heliocidaris erythrogramma*. Grazing urchins cleared many of the plots during the study as the urchin 'barrens' advanced from deeper water into shallower water between late 2013 and mid 2014. This and other studies have demonstrated that *Undaria* is more of an opportunistic species than a competitive invader. Other ecosystem processes have a far greater influence on the composition and cover of the subtidal macroalgal community; these processes may open up space for *Undaria* recruitment and/or aid its growth. Management efforts might be better focused on these influences than that of the ecologically benign *Undaria*. (*The Victorian Naturalist* 137(6), 2020, 258–268)

Keywords: plant-herbivore interactions, invasive species, *Undaria pinnatifida*, *Heliocidaris erythrogramma*

Introduction

Introduced marine species can threaten the biodiversity and ecology of marine ecosystems (Carey *et al.* 2007). Management of introduced species in the marine environment is notoriously difficult and there are few established or effective strategies for doing so. One such problem species is Wakame *Undaria pinnatifida* (Harvey) Suringar (hereafter *Undaria*), a true kelp from the order Laminariales, and native to seas around Japan, Korea and China. *Undaria* has been introduced to temperate ecosystems across the world including: the Atlantic coasts of Western Europe and the south of England (Fletcher and Manfredi 1995; Wallentinus 2007); numerous parts of the Mediterranean (Wallentinus 2007); California (Silva *et al.* 2002); Argentina (Casas and Piriz 1996); New Zealand (Hay and Luckens 1987). *Undaria* was first detected in Tasmania in the late 1980s (Sanderson and Barrett 1989) and in Victoria in 1996. In Victoria, it was found growing on shallow basalt reef near Point Wilson, south-

west Port Phillip Bay (Campbell and Burrage 1998). The genetics of the Victorian population matches *Undaria* from Korea and China, and the introduced population in New Zealand, while the genetics of the Tasmanian population matches *Undaria* from the Sea of Japan (Uwai *et al.* 2007). This suggests that *Undaria* came to Victoria either via New Zealand, or from China/Korea, rather than Tasmania. *Undaria* has since spread throughout Port Phillip Bay, including its Marine Sanctuaries, and to the harbour at Apollo Bay.

Undaria has been characterised as an opportunistic invader of free space (Hay and Villouta 1993; Fletcher and Manfredi 1995; Valentine and Johnson 2004, 2005; Carnell and Keough 2014). While capable of attaining very high densities in exotic environments (especially during the initial phase of its spread in a novel environment), there are conflicting reports about its ecological implications. Studies in Argentina have shown it has no effect on the

density of *Macrocystis pyrifera* beds, nor on associated benthic algae and invertebrates (Rafalo *et al.* 2009), but that it can lead to reduced species diversity in other habitats (Casas *et al.* 2004). Studies in New Zealand have given conflicting reports. Battershill *et al.* (1998) suggested that *Undaria* in Wellington Harbour was causing substantial changes to *Carpophyllum* (Phaeophyceae, Fucales) habitat. Other studies (Brown and Lamare 1994; Hay and Vilouta 1993) provide evidence that *Undaria* is a poor competitor, being confined mostly to habitat otherwise free of algae. In Tasmania, a large number of studies of *Undaria* have found it generally establishes only in areas devoid of other macroalgae, usually urchin 'barrens' formed due to overgrazing by the Short-spined Sea Urchin *Heliocidaris erythrogramma* (Sanderson 1990; Valentine and Johnson 2003; Edgar *et al.* 2004; Valentine and Johnson 2005) or other disturbance of the native algal canopy (Valentine and Johnson 2004).

Evidence from Europe paints it as a benign intruder, though it can become the dominant seaweed in some areas (Wallentinus 2007). It has been shown to be most common in otherwise bare habitat where it has little competition from other algae, and it shows little competitive ability against other large brown algae (Floc'h *et al.* 1996). Fletcher and Farrell (1998) likewise found little adverse ecological impact while pointing out a potentially beneficial role given that it grows in areas otherwise devoid of macroalgae. In California, Thornber *et al.* (2004) found that recently introduced populations were subject to natural controls from local species. Recent work in New Zealand found it had transient effects on the ecosystem, as it showed small effects on community composition but generated a substantial increase in primary production during its annual peak in abundance (South *et al.* 2016).

This project investigated the impact that removal of *Undaria* would have on the native algal canopy in Port Phillip Bay Marine Sanctuaries. Key questions addressed were:

1. Will native algal species occupy space created by the removal of *Undaria*?
2. Following removal of *Undaria*, will occupation of space by native algae reduce its abundance in subsequent seasons?

3. Given limited volunteer resources, what is the most effective frequency and timing for *Undaria* removal?

Methods

The study site was in Jawbone Marine Sanctuary, near Williamstown in the north-west of Port Phillip Bay. Basalt boulder reef forms much of the shoreline of Jawbone Marine Sanctuary and extends 50 to 100 m offshore. The study site was an approximately 20 m wide and 100 m long section of reef facing east-south-east in the middle of the sanctuary (Fig. 1). The depth of the reef within the study site ranged from 0.5 to 2 m below chart datum. The site supported discontinuous macroalgal cover (*Sargassum* spp. dominant) providing space for seasonal *Undaria* growth, and was mostly above the depth of apparently persistent *H. erythrogramma* barrens.

Thirty experimental plots were positioned haphazardly within 5 m either side of a 100 m transect oriented parallel to the shoreline (Fig. 1). Plots were 7 m²—the area within a 1.5 m radius of the subsurface buoy marking each plot. The transect was divided into six consecutive blocks, each containing five plots. Each block was assigned four experimental treatments and one control treatment, with treatments randomly assigned to each plot.

Treatments

Control plots were left undisturbed throughout the experiment. The four experimental treatments consisted of different frequencies and timings of removal of *Undaria*:

1. June and August 2013;
2. June, August and October 2013;
3. August and October 2013;
4. October 2013.

Surveys

Surveys were completed in June, August, October and December 2013, and in March and July 2014. Surveys included counts of all visible *Undaria* thalli in each plot, collection of four 800 mm × 600 mm photoquadrats placed haphazardly in each plot, followed by removal of *Undaria* thalli according to the treatment schedule. Photoquadrats were collected from each plot using a 7 MP digital camera in an underwater housing with a wide-angle lens.

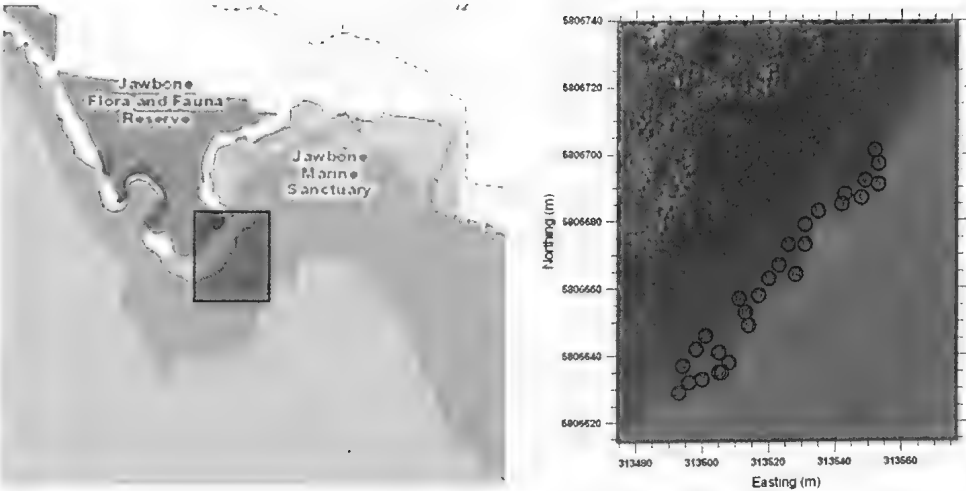


Fig. 1. Left: experimental site in Jawbone Marine Sanctuary (NatureKit). Right: experimental plots (basemap, Nearmap, October 2010, datum WGS84, Zone 55H).

Macroalgal abundance was estimated for each photoquadrat using the point-intercept analysis program Coral Point Count (Kohler and Gill 2006). Thirty-six points were superimposed over each image in a random-stratified array (3 random points in each cell of a 3 × 4 grid covering each image). Each point was assigned

to one of 35 categories, such as the type of red, brown or green alga, the type of invertebrate and the type of bare substrate (Table 1). Where possible, algae were identified to species or genus; when unable to do so, a broad description was provided.

Table 1. Categories used in per cent cover assessment.

Red algae

- Corallina officinalis*
- Encrusting coralline
- Encrusting non-coralline
- Feathery red
- Foliose red
- Geniculate coralline

Brown algae

- Caulocystis cephalornithos*
- Colpomenia sinuosa*
- Cystophora monilifera*
- Cystophora moniliformis*
- Cystophora retorta*
- Dictyota dichotoma*
- Ecklonia radiata*
- Encrusting brown
- Filamentous brown algae
- Laminaria* (kelp) juvenile
- Sargassum* basal leaves
- Sargassum linearifolium*
- Sargassum spinuligerum*
- Scytosiphon lomentaria*
- Undaria pinnatifida*

Green algae

- Caulerpa* spp.
- Codium* spp.
- Ulva* sp.

Invertebrates

- Porifera (Sponge)
- Plesiastraea versipora* (encrusting coral)
- Mytilus* sp. (mussel)
- Coscinasterias muricata* (sea star)
- Heliocidaris erythrogramma* (Shortspined Sea Urchin)
- Tunicata (ascidian or sea-squirt)

Other

- Biotic matrix (sediment/algae/invertebrates)

Bare substrate

- Rock
- Sand
- Sediment
- Shell/Gravel

Data analysis

The mean and variance of the number of *Undaria* thalli within each treatment and survey were calculated from census data and analysed graphically.

The mean and variance of the cover for each category in each plot was computed from photoquadrat data for each treatment and survey. Data for some categories are grouped for presentation including *Sargassum* spp. (both the large annual reproductive thalli and the perennial vegetative basal leaves) and urchin barren (comprising categories associated with barren habitat: bare rock, biotic matrix, encrusting coralline algae, *H. erythrogramma*).

Jawbone Marine Sanctuary urchin barren dynamics

Over the course of the experiment, one of the overwhelming influences on macroalgal cover was the encroachment of *H. erythrogramma* urchin barrens into shallower depths, including into some of our experimental plots. Aerial images clearly showed the movement of the urchin barren 'front'. To gauge the change in the area of the urchin barrens over time, aerial images of the park were imported into Surfer® software and polygons traced over the image. The experimental plots, and the appearance of barren areas within them, were used to ground-truth the aerial images. In July 2014, divers counted the number of *H. erythrogramma* in three 800 mm × 600 mm quadrats placed haphazardly at each of 11 positions distributed evenly along the transect used to place the *Undaria* removal plots (Fig. 1). One quadrat was placed within the barren area, one at the barren edge and one within the intact algal canopy.

Results

The macroalgal community comprised mostly brown algal species with a small number of red and green algal species. *Undaria* was one of the least abundant large macroalgae in 2013, reaching a maximum abundance of just 2.4% cover in spring. On average, *Undaria* accounted for only 0.4% cover. *Sargassum linearifolium* was the dominant macroalga in the study area, accounting for an average of 56% cover over the course of the experiment. It was highly seasonal, reaching a maximum abundance of 84% cover in spring 2013, and a

minimum of 21% cover in autumn 2014. The next most abundant alga was *Ulva* sp. (a green alga) with an average abundance of 7.7% cover. It, too, was highly seasonal, peaking in abundance in autumn 2014 when *Sargassum* cover was minimal. The native kelp *Ecklonia radiata* had an average cover of 1.7%.

Sargassum spp. were not influenced by removal of the small number of *Undaria* thalli in plots, but showed a seasonal trend in abundance, peaking at around 85% cover in October 2013 (Fig. 2a). *Sargassum* cover in two treatments and in the control was much smaller at the end of the study compared to the start. This difference is attributed to the expansion of *H. erythrogramma* barrens into the study area (Fig. 2b). The largest before/after differences in *Sargassum* cover were seen in the control plots and treatment in June, August and October 2013. *Ulva* spp. were present in four of the five surveys, with sporadic changes in its abundance—typical of this species' boom and bust ecology (Fig. 2c). Maximum abundance was seen in early autumn (March 2014) when *Sargassum* cover was at its minimum. It showed minor peaks in abundance in both winter 2013 and winter 2014.

Per cent cover of *Undaria* in the plots peaked in August 2013 (Fig. 2d), but the number of thalli per plot was lower than in June, even in plots where *Undaria* was not removed in June 2013. *Undaria* thalli were largest in August, explaining the peak in cover (August) in spite of having fewer thalli than in June. The number of thalli per plot also was low in October, including in plots where thalli previously had not been removed. These data suggest that, in addition to removal effects, there was some natural attrition in the already small *Undaria* population over the course of removals—which was surprising given that late spring is often the peak for *Undaria* abundance.

Figure 3a shows the average number of *Undaria* thalli counted and subsequently removed from experimental plots for each treatment. The greatest numbers of thalli occurred in June 2013. Figure 3b compares the average number of *Undaria* thalli counted in experimental plots for each treatment and for control plots in June 2013 and at the conclusion of the experiment in July 2014. There were fewer thalli in treatment

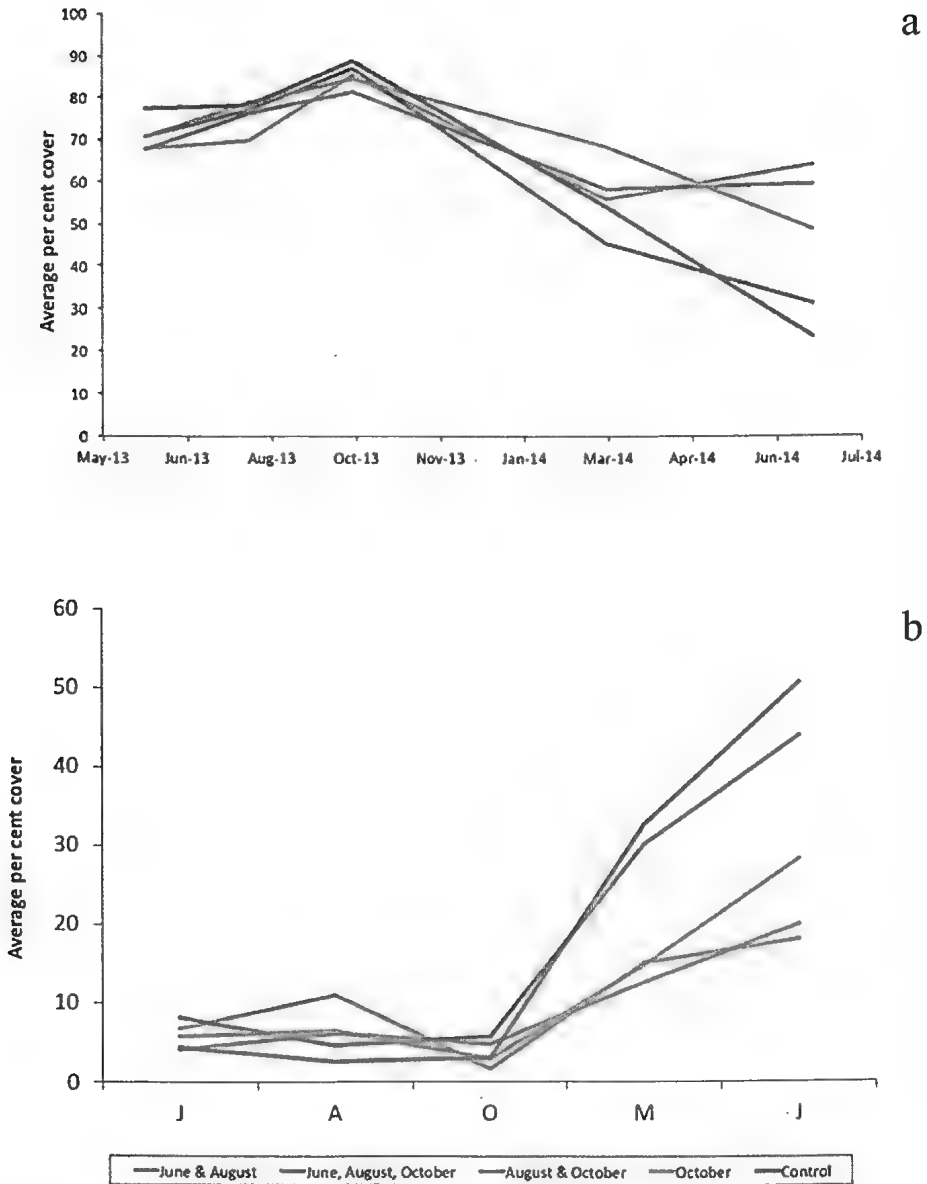


Fig. 2. Average per cent cover in each of the experimental treatment plots, where *Undaria* was removed, and in control plots, for: a) *Sargassum* spp.; b) bare rock. Treatments consisted of different frequencies of removal and timing of removal of *Undaria*: 1. June and August 2013; 2. June, August and October 2013; 3. August and October 2013; 4. October 2013.

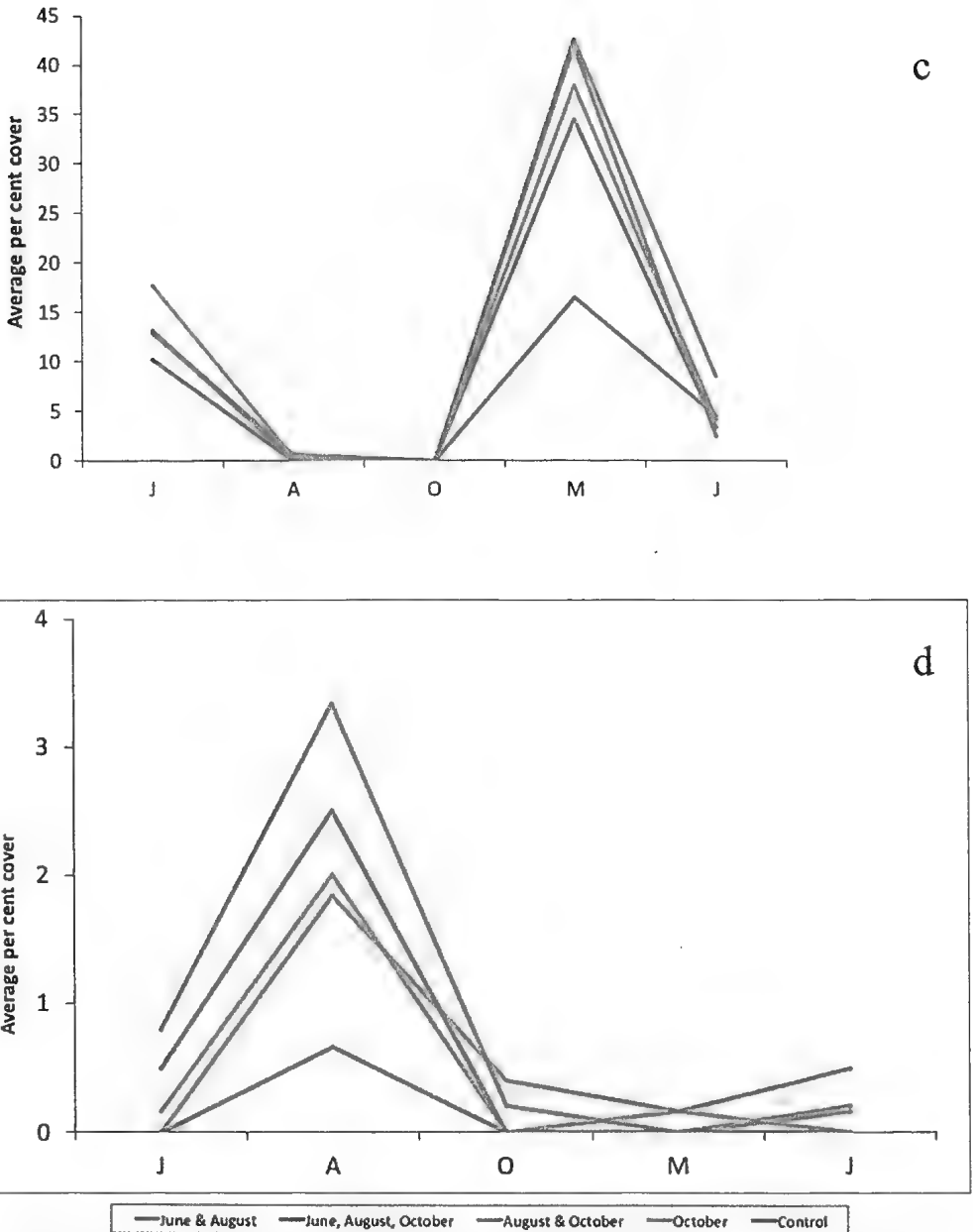


Fig. 2. Average per cent cover in each of the experimental treatment plots, where *Undaria* was removed, and in control plots, for: c) *Ulva* and d) *Undaria*—from June 2013 to July 2014 inclusive. Treatments consisted of different frequencies of removal and timing of removal of *Undaria*: 1. June and August 2013; 2. June, August and October 2013; 3. August and October 2013; 4. October 2013.

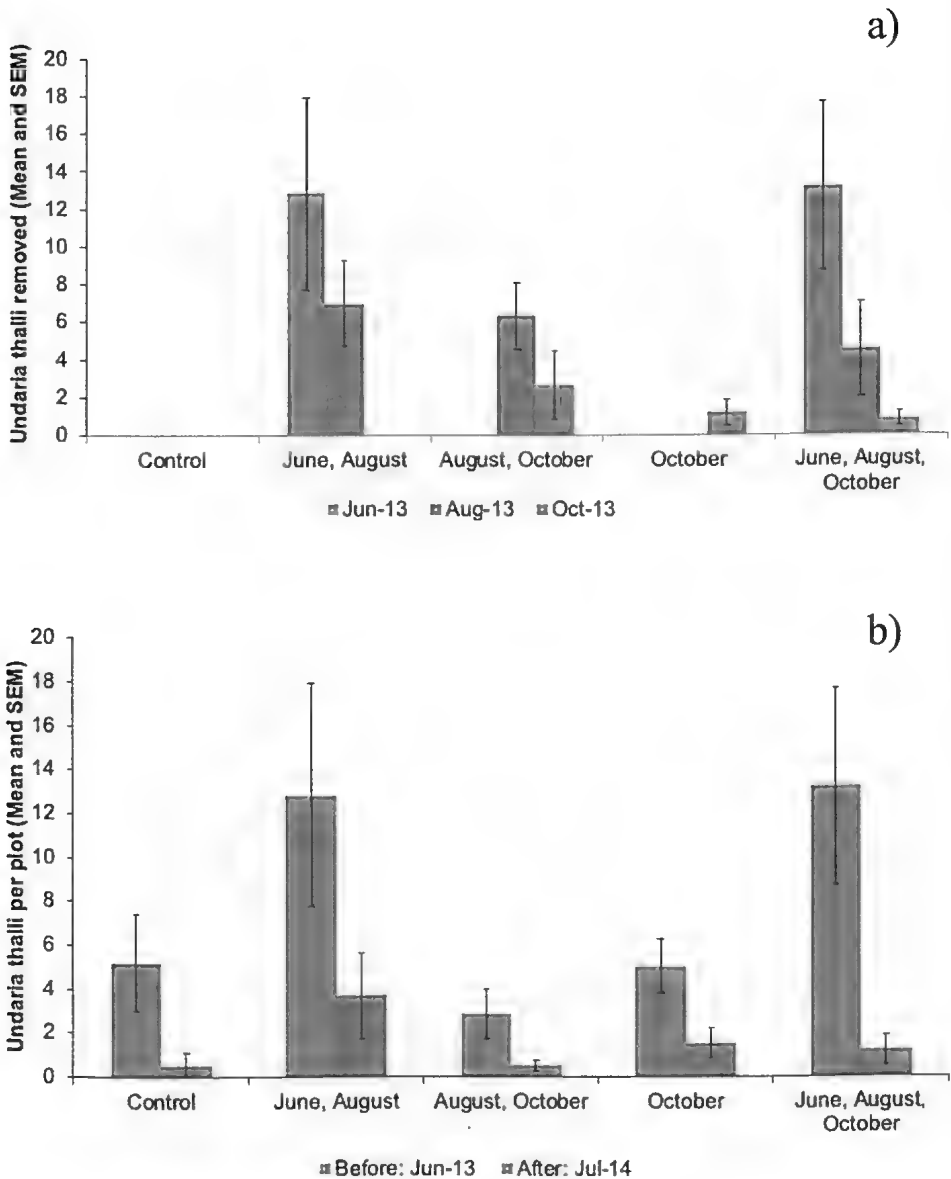


Fig. 3. a) Mean and Standard Error Margin (SEM) of *Undaria* thalli removed per plot from each experimental treatment. b) Mean and SEM of *Undaria* thalli per plot for each experimental treatment and for control plots at the beginning of the experiment (Before: June-13) and in the growing season that followed completion of treatments (After: July-14). Treatments consisted of different frequencies of removal and timing of removal of *Undaria*: 1. June and August 2013; 2. June, August and October 2013; 3. August and October 2013; 4. October 2013.

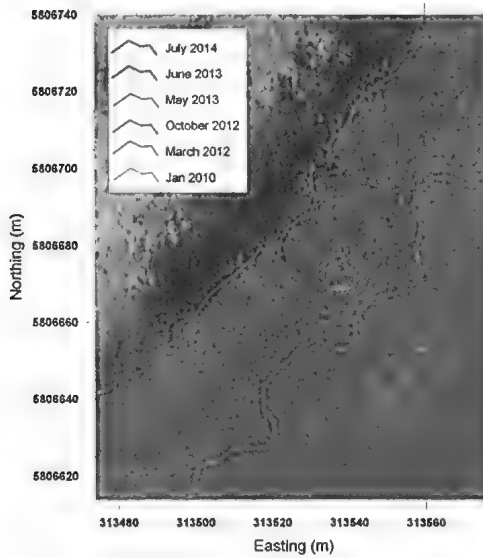
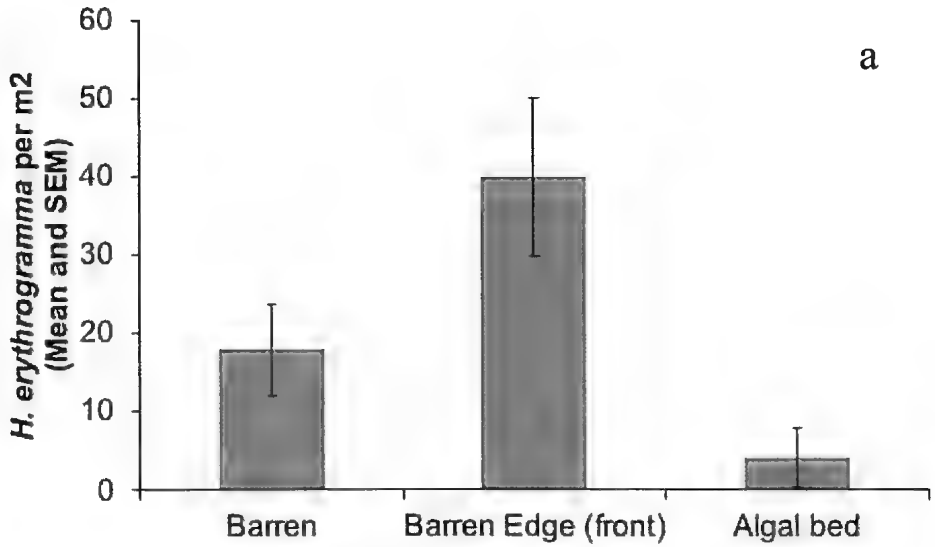


Fig. 4. a) Mean and standard deviation (SD) of *Heliocidaris erythrogramma*, July 2014. b) Shifts in the front of *H. erythrogramma* barren over 2010–2014 period, at the study site.

plots and control plots in July 2014 than in June 2013. The greatest relative changes in numbers of *Undaria* thalli were seen in plots from which *Undaria* was removed in June, August and October (92% decrease) and in controls (90% decrease). There was no difference in the number or cover of *Undaria* thalli between treatment plots and control plots at the end of the study.

Heliocidaris erythrogramma in Jawbone Marine Sanctuary

Figure 4a shows counts of the number of sea urchins in the area of the study in July 2014. The number of sea urchins per square metre was highest at the shallow edge or front of the barren (around 40 urchins per square metre). Numbers within the barren were less than half of this (around 18 per square metre) and the lowest numbers were seen in macroalgae-dominated habitat (3 per square metre). Figure 4b shows a map of the study site, and the border of the algal bed and urchin barren over time. Barrens appear to reach their greatest extent in 2010 and 2014. The extent of barrens in 2014, while noticeably greater than the year before, were not as great as in 2010. Barrens have been persistent in waters over 2 m deep.

Discussion

Understanding the impact of introduced species and the best strategy for managing them is vital for marine conservation. In this study, we found that grazing by the native sea urchin *H. erythrogramma* was overwhelmingly the greatest influence on the native macroalgal community in Jawbone Marine Sanctuary, rather than the introduced kelp *Undaria*. *Heliocidaris erythrogramma* caused a substantial reduction in macroalgal cover as the barren expanded up the depth profile into the study area between spring 2013 and winter 2014. By the final survey in July (winter) 2014, grazing by sea urchins meant that all plots had become urchin 'barrens'. In contrast, the low level of *Undaria* recruitment and growth encountered during the study meant that removals had no effect on the overall cover of macroalgae. Unsurprisingly perhaps, there was no measurable effect of *Undaria* removals on either *Undaria* or native algae. The high existing cover of native

macroalgae in shallow subtidal areas may prevent colonisation by large *Undaria* populations in Jawbone Marine Sanctuary. This finding is consistent with reports from Australia (Valentine and Johnson 2003; Carnell and Keough 2014) and New Zealand (South *et al.* 2016) that demonstrate that intact native algal habitats effectively prevent *Undaria* 'infestation'. For example, the apparent natural attrition of *Undaria* in the spring of 2013 may have been due to the spring growth of *Sargassum* spp. thalli in their reproductive phase, which would reduce available space for *Undaria* recruitment and growth. Intensely grazed urchin barrens likewise showed low *Undaria* density and cover. This finding is in contrast to other studies that have shown *Undaria* sporophytes recruiting in high densities to disturbed habitats, including *Heliocidaris* urchin barrens (Sanderson 1990; Valentine and Johnson 2003, 2004). In Port Phillip Bay, it appears that *H. erythrogramma* may control *Undaria* populations in barren areas, as demonstrated by Carnell and Keough (2016).

There are few examples of removal efforts eradicating or even reducing the abundance of introduced *Undaria* populations. A small number of immature thalli discovered at Flinders in Western Port were removed and *Undaria* has since remained absent (Primo *et al.* 2010). This remains the only known instance of a successful eradication of *Undaria* from a new site. A two-and-a-half year effort to control *Undaria* in the Tinderbox Marine Reserve near Hobart in Tasmania was able to reduce the abundance of *Undaria*, but required substantial effort (monthly removals) (Hewitt *et al.* 2005). Despite removal efforts, 'hotspots' of *Undaria* growth remained, presumably due to persistent gametophyte populations (Hewitt *et al.* 2005). Underwater operations are always time consuming and expensive—the removal of *Undaria* for this study was no exception. The costs of *Undaria* removal, except where there is a high confidence that a population is new and of a known and limited size, are unlikely to justify the result. While Parks Victoria has identified that there is a risk of introduced species causing changes to ecology in marine parks and sanctuaries (Carey *et al.* 2007), there is little evidence

that *Undaria* has led to changes in ecology per se. It is characterised more as an opportunistic species, albeit a highly fecund one, which, in ecological terms, appears to be relatively benign (Wallentinus 2007; Fletcher and Farrel 1998; Floe'h *et al.* 1996).

Undaria is known to almost exclusively recruit to bare substrate devoid of other macroalgae (Valentine and Johnson 2003, 2004; Carnell and Keough 2014). In the case of *Undaria*, it seems that the ecological change takes place before the actual invasion (Valentine and Johnson 2003, 2004, 2005). Furthermore, Valentine and Johnson (2005) found that recovery of the native algal canopy was not facilitated simply by removal of *Undaria* and that factors such as depth, grazing and sedimentation were just as important. Indeed, the combination of disturbance to the native canopy and addition of nutrients can lead to a synergistic increase in *Undaria* recruitment and cover (Carnell and Keough 2014).

The key feature of *H. erythrogramma* populations in Jawbone Marine Sanctuary, and the barrens they create on reefs throughout Port Phillip Bay, is the movement of their upper depth limit. *Heliocidaris erythrogramma* barrens are typically confined to water at least one metre below low water mark, but this limit may be shallower as demonstrated by aerial images from Jawbone Marine Sanctuary. These images also show that *H. erythrogramma* maintains persistent barren areas in water over 2 m deep throughout the park. More broadly, in Port Phillip Bay, *H. erythrogramma* forms barrens in a variety of habitats, including reefs (Carnell and Keough 2019), seagrass beds (Constable 1989) and soft sediment *Pyura dalbyi* (Crockett 2012).

There may be bottom-up and top-down controls on the upper depth limit of *H. erythrogramma* barrens. Ling *et al.* (2010) suggested *H. erythrogramma* forms barrens only in areas of low water movement. Deeper water would provide a refuge from the effects of wave action in Port Phillip Bay, exerting a bottom-up control. Livore and Connell (2012) suggest the lower nutritional quality and availability of kelp in

sheltered locations could drive creation of urchin barrens. Studies in Port Phillip Bay suggest decreases in nutrients and, therefore, primary productivity, may cause the urchins to shift to grazing attached algae (Carnell and Keough 2019). Vulnerability to predation by seabirds, such as the Pacific Gull *Larus Pacificus georgii* (Keesing 2001), may exert a top-down control, discouraging urchins from entering shallow water. Additionally, lower salinity in shallow water may drive stenohaline echinoderms such as *H. erythrogramma* into deeper water in wet periods. There were reports of drastic reductions in numbers of the North Pacific Seastar *Asterias amurensis*, especially in the Maribyrnong River, at the end of the Millennium Drought (J Lewis, pers. comm. July 2013). Few *H. erythrogramma* were found in shallow water in this study. When seen, they were typically hiding between rocks or beneath the macroalgal canopy. The edge of the barrens, visible in images in 2003 and 2009 (during the period of prolonged drought), however, appears to show them extending to the intertidal boundary.

The data collected here demonstrates the importance of understanding the threat posed by marine introduced species, and the drivers of their abundance, before investing significantly in their management. From this case study and others, it is apparent that *Undaria* is an opportunistic species rather than a competitive invader. Other ecosystem processes have a far greater influence on the composition and cover of the subtidal macroalgal community; these processes may open up space for *Undaria* recruitment and/or aid its growth. Therefore, managing sub-tidal reef ecosystems to promote a healthy native algal canopy should be the primary focus, rather than removal programs of *Undaria*.

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Human-mediated dispersal of the seeds of Australian weeds

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Abstract

Human-mediated seed dispersal facilitates biological invasions, including the spread of noxious and/or naturalised alien plants in Australia—but which species are dispersed by such mechanisms and what traits do they share? Data were extracted from two reviews of seed dispersed from clothing/personal equipment (e.g. backpacks) and vehicles to assess the range and types of Australian weeds benefiting from these dispersal mechanisms. Across 33 studies, including 11 from Australia, 466 species of weeds in Australia were recorded, 375 from vehicles and 212 from clothing. Many noxious species (17%) and ten Weeds of National Significance were included. Species were predominantly forbs or graminoids with few shrubs or trees. Seeds from annuals were more frequent than expected on vehicles but perennials were more frequent on clothing. New studies will add species to this corpus, but the results already highlight the importance of strategies to minimise risks from such dispersal mechanisms to help reduce the homogenisation of floras, including those in Australia. (*The Victorian Naturalist* 137 (6), 2020, 269–275)

Keywords: biological invasion, long distance dispersal, invasive species, vehicles, clothing

Introduction

Weeds, native and introduced plants growing outside their natural range and unwanted and/or naturalised, are a major issue in Australia (Coleman *et al.* 2011). They cost billions of dollars from lost productivity in agricultural areas (Invasive Plants and Animals Committee [IPAC] 2016), are a major threat to native biodiversity including in protected areas (Coleman *et al.* 2011), have altered fire-prone regions, and are changing coastal and riverine systems (IPAC 2016; Richardson *et al.* 2016). It is estimated that there are over 28 000 species of plants introduced into Australia, 3200 of which are naturalised and more than 500 species declared noxious by government authorities. Many of these are highly invasive (Coleman *et al.* 2011; IPAC 2016), with 32 species listed as Weeds of National Significance (IPAC 2016).

A key step in biological invasions is dispersal, with long distance dispersal contributing to the rapid spread of species (Bullock *et al.* 2018; Sinclair *et al.* 2019). Among the many dispersal mechanisms for weeds, there is increasing recognition of the importance of human-mediated seed dispersal, particularly when it results both in long distance dispersal and high propagule loads (Bullock *et al.* 2018; Sinclair *et al.* 2019). Reviews of invasive alien species highlight how escalating human movement combined with

rapid climate change and habitat alteration results in suites of similar species dominating similar disturbed habitats globally (Bullock *et al.* 2018), causing homogenisation of floras in various locations and regions (McNeely 2001; Olden *et al.* 2004). Such patterns, including the dominance of similar weed floras in different regions, are also seen in Australia (Richardson *et al.* 2016).

Previous reviews have documented the diversity of species, including weeds that can be dispersed by humans deliberately for horticulture, agriculture and forestry, as well as through unintentional human-mediated dispersal (McNeely 2001; Randall 2007; Coleman *et al.* 2011; IPAC 2016; Bullock *et al.* 2018; Sinclair *et al.* 2019). For example, a global review based on data from 13 studies (Ansong and Pickering 2013) found that seeds from 626 species can be dispersed unintentionally from vehicles, with 599 of these species considered weeds somewhere in the world. When 21 studies on seed dispersal from clothing were reviewed, a wide diversity of species was found to be spread unintentionally, both near and far. Seed from 449 species were recorded, with 391 considered weeds somewhere in the world (Ansong and Pickering 2014).

To highlight the role of human-mediated dispersal for weeds in Australia, data were extracted and combined from a review of seed dispersed from vehicles and a review of seed dispersed from clothing respectively (Ansong and Pickering 2013, 2014). This enabled an assessment of the diversity and types of weeds in Australia that can be dispersed from clothing and vehicles, the characteristics of these weeds and the results of studies conducted in Australia. The implications of the results were also addressed, which included ways of minimising seed spread by these methods of dispersal and highlighted the need for further research, including in Australia.

Methods

Data concerning each species considered a weed in Australia were extracted from the above-mentioned reviews (i.e. Ansong and Pickering 2013, 2014). Data included scientific and common name, family, which and how many studies the weed was recorded in, the type of weed (naturalised, noxious and/or Weed of National Significance) and its growth form and life form. Species were considered to be a weed in Australia if they were listed as naturalised in Randall (2007) and/or if they were listed as noxious weeds by a state or territory government (Australian Government 2012). Data about any species recognised as Weeds of National Significance by the Australian Government (IPAC 2016; Australian Government 2020a) were also recorded. The extracted information about Australian weeds was then summarised in tables and using Chi-squared tests in Excel, to show if species with specific traits were more, or less, likely to be dispersed by one or other of the two mechanisms (i.e. seeds dispersed from vehicles or clothing). Data were adjusted for the total number of species per method of dispersal.

Results

There were 33 publications listing species dispersed by these two types of human-mediated seed dispersal. Thirty-two studies included one or more species considered weeds in Australia. Eleven studies were from Australia (Table 1). Seven of these examined seed on clothing and equipment, three from New South Wales, one from Queensland, one from a sub-Antarctic

island, one from the Northern Territory, and one where there was limited detail provided about where the clothing was worn (Wace 1985). Collectively, 106 species of Australian weeds were recorded in these studies. Four studies examined seed dispersed by vehicles and included studies carried out in the Australian Capital Territory, Northern Territory, Queensland, New South Wales and Victoria. Collectively, 270 species of Australian weeds were recorded for these states and territories.

As the number of studies in Australia and overseas increased, the cumulative number of Australian weeds known to be dispersed from clothing and vehicles also increased, indicating that there are still far more species dispersed by these two mechanisms than have been documented so far. Some studies, such as those examining material from a large number of vehicles in Australia (e.g. Wace 1977), have recorded both high diversity within the study and added many new species to the total corpus. Wace (1977) germinated 68 species from mud collected from a carwash; Moerkerk (2006) recorded 178 species of Australian weeds amongst seed germinating from material washed down from a wide range of vehicles in south-eastern Australia; a study looking at material collected from seed traps along motorway tunnels in Berlin, Germany (Von der Lippe and Kowarik 2007, 2008) recorded 134 species, of which 96 are weeds in Australia. Similarly, a large study of seed collected from footwear and clothing of people cutting plants in 47 different meadows in Sweden recorded 197 species, 96 of which are weeds in Australia (Auffret and Cousins 2013).

Across all studies, seed from 466 species that are weeds in Australia were recorded. These were mainly from vehicles (375 species), but many also were recorded from clothing and equipment such as backpacks (212 species) (Table 2). Nearly all species were naturalised in Australia (99%), with 17% considered noxious and 5.2% invasive. Sixty-five families of Australian weeds were represented, with 60 families being from vehicles (Table 2). The Poaceae was most common (128 species), followed by the Asteraceae (63 species), Fabaceae (39), Brassicaceae (32) and Caryophyllaceae (22).

Table 1. Details of 11 studies in Australia examining human-mediated seed dispersal on clothing and equipment or vehicles identified in Ansong and Pickering (2013, 2014).

Zone: Subtrop. = Subtropical; Temp. = Temperate; MT = mountain; Trop. = Tropical.
 Study types: O = Opportunistic sampling; NE = Natural experiment; ME = Manipulative experiment.
 Vector: CE = clothing and equipment.

Reference	Zone	Study Type	Vector	Location	Taxa	Australian Weeds
Pickering and Mount 2008–2009	Subtrop.	ME	CE	Roads and trails, Gold Coast, QLD	-	29
Whinam <i>et al.</i> 2005	Various	NE	CE	Macquarie Island, Sub-Antarctic Islands	81	24
Pickering and Mount 2009	Temp./MT	ME	CE	Kosciuszko National Park, NSW	-	21
Wace 1985	Various	O	CE	Limited details	38	20
Mount 2008	Temp./MT	ME	CE	Kosciuszko National Park, NSW	70	19
Pickering <i>et al.</i> 2011	Temp.	ME	CE	Kosciuszko National Park, NSW	5	3
Scott 2009	Trop.	O	CE	Darwin, Northern Territory	1	1
Moerkert 2006	Temp.	NE	Vehicle	South-eastern Australia (NSW, SA, Vic)	222	178
Wace 1977	Temp.	NE	Vehicle	Canberra, ACT	259	68
Lonsdale and Lane 1994	Trop.	NE	Vehicle	Kakadu National Park, NT	88	11
Nguyen 2011	Subtrop.	NE	Vehicle	Rural road QLD	146	63

Most species were forbs (279) or graminoids (143), with fewer species being shrubs (25) or trees (19). There was no significant difference in the relative diversity of forbs, graminoids or trees between the two dispersal vectors, although, with further sampling, differences in shrubs may be revealed as only five species of shrubs were recorded from clothing compared to 22 from vehicles (Table 2). There was similar diversity of annual (190 species) and perennial (197) weed species overall, but annual species were more likely to be found on vehicles than clothing, which was unexpected (Chi-squared <0.05). For perennials, it was the reverse; they occurred disproportionately on clothing (Table 2).

Eight species were recorded in 10 or more of the studies, with the small Annual *Poa Poa annua* the most common, being recorded in 16 studies (Table 3). Australian weeds common among the studies are well recognised weeds in many parts of the world and, while important, are not considered to be of national significance in Australia. However, there were ten Weeds of National Significance in Australia, nine dispersed by vehicles (including two species of *Salix*), and two, namely Serrated Tussock *Nassella neesiana* and Gamba Grass *Andropogon gayanus*, dispersed by clothing.

Discussion

Human-mediated seed dispersal from vehicles and equipment and clothing are likely to be important for many Australian weeds. Seeds on vehicles can contribute to the spread of Australian weeds in urban (Wace 1977), agricultural (Nguyen 2011; Bajwa *et al.* 2018; Khan *et al.* 2018) and protected areas (Lonsdale and Lane 1994). Research has already shown that the quantity and diversity of weed seeds on vehicles varies with the type of vehicle, where it is driven, how clean it is and, importantly, when it is driven (Ansong and Pickering 2013; Nguyen 2011; Bajwa *et al.* 2018; Khan *et al.* 2018; Rew *et al.* 2018). Generally, there are more seeds on four-wheel drive vehicles, on vehicles driven off road, on vehicles driven when plants are seeding in spring, summer or autumn (Khan *et al.* 2018; Bajwa *et al.* 2018), and on vehicles with mud attached, particularly to the wheels, wheel rims and underside (Khan *et al.* 2018).

Table 2. Number of Australian weed species found on clothing, vehicles and overall, with different traits. Significant Chi-squared tests are in bold.

Characteristics	Clothing	Vehicle	All	Chi-square Test
Weed status in Australia				
Naturalised in Australia	211	370	461	0.874
Invasive in Australia	7	21	24	0.225
Weeds of National Significance	2	9	10	
Noxious in Australia	28	64	79	0.266
All families	40	60	65	
Poaceae	71	101	128	
Asteraceae	27	53	63	
Fabaceae	16	32	39	
Brassicaceae	9	28	32	
Caryophyllaceae	15	15	22	
Cyperaceae	4	8	11	
Apiaceae	7	5	10	
Lamiaceae	3	9	10	
Malvaceae	1	10	10	
Growth form				
Graminoid	78	112	143	0.147
Forb	121	226	279	0.661
Shrub	5	22	25	0.058
Tree	8	15	19	0.903
Life form				
Annual	69	164	190	0.0423
Perennial	108	145	197	0.0267
Annual/Biennial	11	27	30	0.3651
Annual/Perennial	12	30	32	0.3159
Biennial	7	6	10	0.1801
Biennial/Perennial	5	3	7	

Recommendations for ways to reduce weed seeds dispersed by vehicles, particularly four-wheel drive and other vehicles used off-road, include washing down vehicles entering areas such as parks, or where specific weeds are of concern in agricultural areas (Bajwa *et al.* 2018; Khan *et al.* 2018; Rew *et al.* 2018).

Although fewer species of Australian weeds were recorded on clothing and equipment than on vehicles, clothing and equipment remain of concern because they can act as mechanisms for very long distance dispersal, and can introduce large numbers of seeds into remote natural areas (Pickering *et al.* 2011), even sub-Antarctic islands (Whinam *et al.* 2005) and Antarctica (Chown *et al.* 2012; Hughes *et al.* 2020). As a result, protocols have been introduced for people venturing into remote areas of high conservation value to ensure that clothing and equipment is clean (Whinam *et al.* 2005; Hughes *et al.* 2020). In some cases there are preferences for certain types of material (Whinam *et al.* 2005), with seeds more

likely to remain attached for longer on fleecy materials compared to material with smoother, tighter weaves (Ansong *et al.* 2015; Ansong and Pickering 2016). Similarly, the types of boots worn can affect the amount of seeds collected. Those with deep tread where soil accumulates are potentially able to disperse more seed over longer distances than other types of footwear (Wichmann *et al.* 2009). The risk of clothing and equipment as a means of dispersing seed into countries is already recognised (McNeill *et al.* 2011). Shoes and clothing contaminated with soil or plant material are required to be declared when entering Australia to minimise introductions of new species (Australian Government 2020b).

Despite the importance of human-mediated seed dispersal, there are relatively few studies on this topic worldwide and in Australia. This was true not only at the time of the reviews but remains the case. In Australia, there appear to be only a handful of more recent studies, including:

Table 3. Characteristics of the Australian weeds most often recorded across the studies and Weeds of National Significance dispersed by one or more studies of dispersal from vehicles or clothing.

Species	Family	Common Name	Geographic Origin	Growth Form	Life Form	All	Vehicle	Clothing
Most common species across 33 studies								
<i>Poa annua</i>	Poaceae	Annual Poa	Europe	graminoid	annual	16	6	10
<i>Poa pratensis</i>	Poaceae	Kentucky Bluegrass	Europe, N Africa, Asia, N America	graminoid	perennial	12	7	5
<i>Trifolium repens</i>	Fabaceae	White Clover	Europe, N Africa, Temp Asia	forb	perennial	11	7	4
<i>Holcus lanatus</i>	Poaceae	Yorkshire Fog	Europe, N Africa, Temp Asia	graminoid	perennial	11	4	7
<i>Dactylis glomerata</i>	Poaceae	Cocksfoot	Europe, N Africa, Asia	graminoid	perennial	11	5	6
<i>Plantago major</i>	Plantaginaceae	Large Plantain	Europe, Asia	forb	perennial	10	6	4
<i>Lolium perenne</i>	Poaceae	Perennial Ryegrass	Europe, N Africa, Asia	graminoid	perennial	10	5	5
<i>Taraxacum officinale</i>	Asteraceae	Dandelion	Europe	forb	perennial	10	5	5
Weeds of National Significance								
<i>Andropogon gayanus</i>	Poaceae	Gamba Grass	Africa	graminoid	perennial	1		1
<i>Chrysanthemoides monilifera</i>	Asteraceae	Bitou Bush	S Africa	shrub	perennial	1	1	
<i>Genista linifolia</i>	Fabaceae	Mediterranean Broom	Europe, North Africa	shrub	perennial	1	1	
<i>Nassella neesiana</i>	Poaceae	Chilean Needle Grass	S America	graminoid	perennial	1	1	
<i>Nassella trichotoma</i>	Poaceae	Serrated Tussock	S America	graminoid	perennial	2	1	1
<i>Parthenium hysterophorus</i>	Asteraceae	Parthenium Weed	Caribbean region, possibly parts Central, N and S America	forb	annual	1	1	
<i>Rubus fruticosus</i>	Rosaceae	Shrubby Blackberry	Europe	shrub	perennial	2	2	

Table 3. Continued

Species	Family	Common Name	Geographic Origin	Growth form	Lifeform	All	Vehicle	Clothing
<i>Salix caprea</i>	Salicaceae	Pussy Willow	Europe, Central Asia	shrub/tree	perennial	2	2	
<i>Salix triandra</i>	Salicaceae	Almond Willow	Europe, Western and Central Asia	shrub/tree	perennial	1	1	
<i>Ulex europaeus</i>	Fabaceae	Gorse	Europe	shrub	perennial	1	1	

experimental studies examining seed on clothing, quantifying how far seed can be carried and then dispersed from clothing (Ansong *et al.* 2015); how long seed with different structures remains attached to specific fabrics (Ansong and Pickering 2016); modelling dispersal of seed over distance by hikers (Pickering *et al.* 2011); and unpublished data comparing seed attaching to a mountain biker and a hiker in rural and natural areas (Pickering and Mount 2008–2009, unpubl. data 2009). For vehicles, there have been a few new studies. These include one study assessing seed germination from material collected from four-wheel drive vehicles at different times of the year in Queensland. Ninety-one species were recorded with 60 not native to Australia (Khan *et al.* 2018). Another Queensland study was of seeds germinated from material collected from vehicle wash-downs in agricultural areas. One hundred and forty-five species were identified, of which at least 37 were introduced into Australia, and included *Parthenium Weed Parthenium hysterophorus*, a Weed of National Significance (Bajwa *et al.* 2018).

Additional research in Australia and internationally is likely to continue to identify more species that have seed dispersed by one or more of these means, including species that are weeds in Australia. For example, the research on seeds on vehicles in Australia (Khan *et al.* 2018; Bajwa *et al.* 2018), the USA vehicles used by the US Army (Rew *et al.* 2018) and China (vehicles entering a mountain national park) (Yang *et al.* In Press) have continued to find a diversity of species, and include species not recorded in the review by Ansong and Pickering (2013). Similar research on seeds on clothing, such as a recent study of people flying into New Zealand, has recorded taxa new to global records (McNeill *et al.* 2011), while experimental research has investigated factors affecting potential seed dispersal on boots (Hardiman *et al.* 2017). With new studies identifying novel weed species being dispersed, there is a clear need for further research on these two types of dispersal mechanisms, including from more regions and situations, as well as for other types of human-mediated seed dispersal such as from mountain bikes (Pickering *et al.* 2016). Such studies will continue to increase our understanding of the importance of human-mediated seed dispersal of weeds in Australia and their contribution to the homogenisation of the flora. Further research is also necessary to assess means and processes for minimising the risk posed by these dispersal mechanisms, including the effectiveness of strategies such as shoe and vehicle cleaning and educating people about the risk of spreading weed seed in areas of high conservation value (Ansong and Pickering 2015).

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Impacts of introduced deer in Victoria

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Abstract

Four species of introduced deer have established wild populations in Victoria. These populations can exert negative impacts on biodiversity, agriculture and public health, particularly where they occur at high densities. Biodiversity impacts primarily relate to the direct effects of grazing/browsing, wallowing and trampling on the structure and composition of indigenous floral communities, possibly resulting in indirect impacts on fauna and ecosystem processes. Agricultural impacts include damage to crops and competition with livestock, damage to infrastructure and potential transmission of livestock diseases. Public health impacts include vehicle collisions and transmission of zoonotic pathogens in drinking water catchments. We provide a brief review of these impacts in Victoria and show that they are widespread across the state. While the existing evidence is limited, this is an active field of research, and comparative and experimental studies will help to address knowledge gaps and assist with the development of strategies for managing the impacts of deer in Victoria. (*The Victorian Naturalist* 137 (6), 2020, 276–281)

Keywords: agriculture, biodiversity, public health, introduced deer

Introduction

Globally, deer have been widely introduced to new ranges, where they have established populations in a variety of habitats (Long 2003). In Australia, establishment of wild deer began in the mid-1800s when Acclimatisation Societies released deer for hunting and continued with illegal translocations, accidental farm escapes, and deliberate releases (Moriarty 2004). Wild deer are now present in every Australian state and territory and occupy a range of habitats (Davis *et al.* 2016). Victoria currently has wild populations of four deer species: Sambar Deer *Rusa unicolor* (also sometimes referred to as *Cervus unicolor*); Fallow Deer *Dama dama*; Red Deer *Cervus elaphus*; and Hog Deer *Axis porcinus*. Evidence shows that the distributions and abundances of these populations and the magnitude of recreational harvest, particularly of Sambar and Fallow Deer, have increased dramatically in recent years (Forsyth *et al.* 2018). Deer occur across most of the state, and populations continue to expand—e.g. Sambar Deer at Wilsons Promontory National Park (NP).

Deer can be keystone species and ecosystem engineers due to their ability to modify ecosystem function at a landscape scale (Côté *et al.* 2004). Detrimental effects on natural and production ecosystems have been documented

globally but are less well studied in Australia (Davis *et al.* 2016). Nonetheless, evidence has been deemed adequate to support national listing of 'Herbivory and environmental degradation caused by feral deer' as a Key Threatening Process under the *Environment Protection and Biodiversity Conservation Act* 1999 (Australian Government Department of Agriculture, Water and the Environment 1999) and 'Reduction in biodiversity of native vegetation by Sambar (*Cervus unicolor*)' as a Potentially Threatening Process under the *Victorian Flora and Fauna Guarantee Act* 1988 (Department of Sustainability and Environment 2010). While wild deer can have positive social and economic impacts, for example for regional communities through recreational hunting (Knoche and Lupi 2012) and commercial harvesting, we focus on negative impacts. We draw on the review of deer ecology, impacts and management by Davis *et al.* (2016) and research since 2016 to provide a brief summary of the known negative impacts of deer in Victoria.

Biodiversity impacts

Wild deer exert a range of negative impacts on biodiversity values in Victoria, primarily resulting from the direct effects of herbivory,

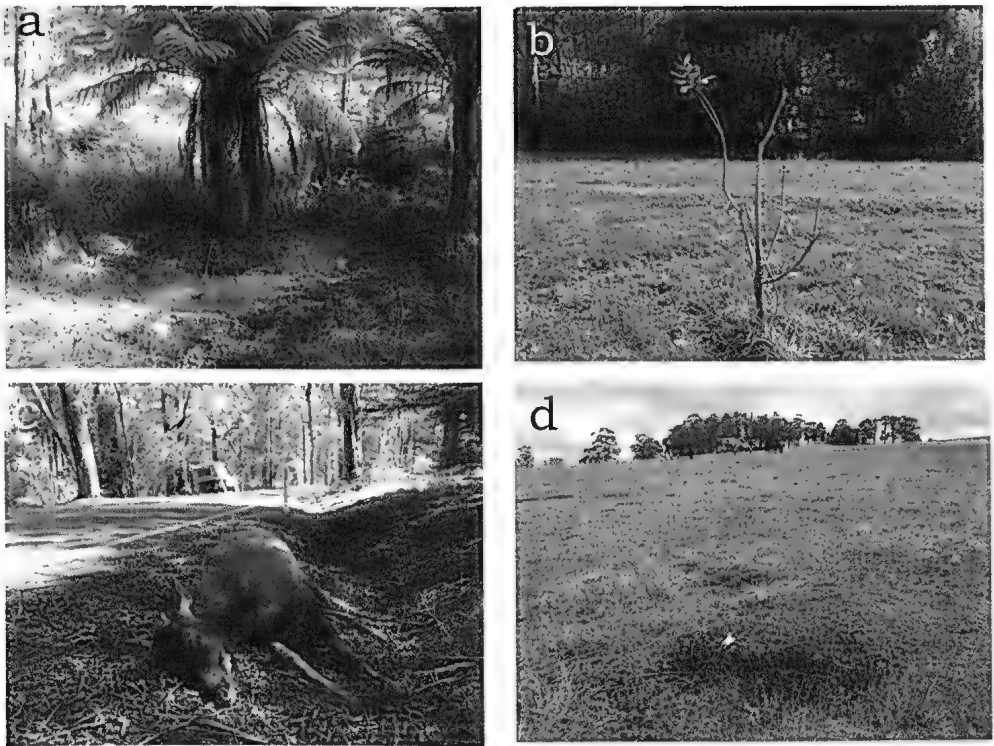


Fig. 1. Biodiversity, agricultural and public health impacts of introduced deer in Victoria: a) Sambar Deer wallow in a national park; b) browsing damage by Fallow Deer to a fruit tree on a farm; c) Sambar Deer after collision with a vehicle; d) potential disease spread to livestock from Sambar Deer faecal pellets on a farm. Photos J Hampton.

antler rubbing and thrashing, trampling and wallowing (Fig. 1a) on indigenous flora (Davis *et al.* 2016). Here we summarise key evidence from Victorian studies that suggests biodiversity impacts are widespread across the state, with the potential for cumulative damage resulting from multiple deer species, which can occur in sympatry in habitats ranging from coastal grasslands to forests to alpine peatlands (Davis *et al.* 2016).

Deer can reduce plant growth, survival and reproduction, with consequences for community composition and structure, and persistence of species and communities. Impacts are pronounced when plant species and size classes are selectively targeted (e.g. palatable browse species, or species with characteristics favoured for rubbing) and are vulnerable to these activities (e.g. have a low tolerance of browsing or trampling), with potentially

severe consequences for rare species. Similarly, community-level impacts on biodiversity are pronounced for vegetation communities that are selectively used by deer, spatially restricted and vulnerable to deer damage (e.g. alpine peatlands favoured for wallowing). Further, changes in vegetation community structure and composition can have indirect impacts on fauna and ecosystem processes, for example, by altering habitat suitability and water quality, impacting on processes such as nutrient cycling and interacting with disturbances such as fire. Peel *et al.* (2005) provide a detailed narrative describing the mechanisms by which deer impact on the natural environment, with a focus on Sambar Deer in East Gippsland.

In Victoria, direct impacts of deer herbivory have been quantified in a range of vegetation types and geographic locations. In central Victoria, Sambar Deer herbivory reduces plant

biomass in the shrub layer and reduces vertical growth in forests of the Yarra Ranges NP (Bennett 2008). In riparian zones in this park, as well as Yarra Ranges State Forest, Bunyip State Park, Tarago State Forest and surrounding peri-urban and agricultural areas, deer browsing impacts increase as deer density increases (Hazel *et al.* 2020). In these riparian areas, deer density is highest at low elevations near large waterbodies where tree cover is intermediate to high and, for a given deer density, impacts are greatest at warmer, wetter sites with an easterly or southerly aspect. In the Grampians NP in western Victoria, Red Deer reduce vegetation cover and tree regeneration in woodland (Roberts 2013) and at Wilsons Promontory NP in the south-east of Victoria, Hog Deer contribute to reductions in plant biomass in coastal grassy woodland (Davis 2014). Increasing levels of deer activity in peatland vegetation communities suggest that such impacts are also occurring in alpine locations in Victoria's north-east, such as the Alpine NP (Batpurev 2017). Impacts of herbivory, such as reduced growth and physical damage, also have been documented on revegetation plantings at Wilsons Promontory NP (Davis and Coulson 2010) and in central Victoria at Yellingbo Nature Conservation Reserve (Moser and Greet 2018). Importantly, antler rubbing by Sambar Deer impacts on threatened species in Victoria, damaging and killing plants. For example, Sambar Deer in the Yarra Ranges NP damage Shiny Nematolepis *Nematolepis wilsonii*, selectively using saplings with a large stem diameter for antler rubbing (Bennett and Coulson 2011). In East Gippsland, antler rubbing by Sambar Deer impacts on several threatened species such as Buff Hazelwood *Symplocos thwaitesii* (Peel *et al.* 2005) and Yellow-wood *Acronychia oblongifolia* (Bilney 2013). Such impacts may be pronounced following fire, when deer can be attracted to regrowth in burnt areas (Downes 1983; Forsyth *et al.* 2012), with likely destructive impacts on post-fire regeneration.

There is also evidence that deer in Victoria have indirect impacts on vegetation communities. Multiple deer species are likely to act as seed dispersers for exotic and native plant species in diverse habitats across Victoria, given that

viable seeds are consumed and defecated by Sambar Deer in the forests of Mount Buffalo NP in the north-east (Eyles 2002) and Hog Deer in coastal grassy woodland at Wilsons Promontory NP (Davis *et al.* 2010). Further, Fallow Deer and Sambar Deer consume fungi in pine plantations near Lake Eildon NP and Toombulup State Forest along the eastern boundary of Mount Samaria State Park, north-eastern Victoria (Parker 2009), possibly altering spore dispersal patterns. Activities such as wallowing, trampling, rutting and fighting also can impact on vegetation communities by removing native vegetation (e.g. Sambar Deer in central Victoria, at Lake Mountain, Yarra Ranges NP, and Mount Bullfight Nature Conservation Reserve; Bennett 2012), which can cause erosion and facilitate weed invasion.

Evidence for the impacts of deer on Victorian fauna is limited. It is likely that modification of vegetation by deer reduces habitat suitability for some Victorian fauna. Bartlett (2012) found that sites in the Yarra Ranges NP with high Sambar Deer densities were associated with reductions in small mammal species richness, abundances of some small mammals and reptile captures, possibly due to reductions in availability of shelter, food and nesting resources. In addition, it is likely that deer compete with native fauna for food and habitat throughout their Victorian range, given overlap in diets and habitat use between native herbivores and Hog Deer in coastal habitats in the south-east and east of Victoria at: Wilsons Promontory NP (Davis *et al.* 2018), Sunday Island, Blond Bay State Game Reserve and Gippsland Lakes Coastal Park (Davis 2013); Sambar Deer in forested habitats in central Victoria in the Yarra Ranges NP and surrounds (Forsyth and Davis 2011); and Red Deer in woodland habitats in western Victoria at the Grampians NP (Roberts *et al.* 2015). Further, deer are eaten by Wild Dogs *Canis familiaris*, Dingoes *Canis dingo* and European Red Foxes *Vulpes vulpes* in Victoria, which may influence predator population dynamics and their use of native prey species (Forsyth *et al.* 2018, 2019). Deer also may have indirect effects on predation by opening the understorey and facilitating predator movement (Davis *et al.* 2016).

Agricultural impacts

Wild deer exert a range of negative impacts on agriculture (Putman and Moore 1998; Finch and Baxter 2007). Across Victoria, these impacts include direct damage to crops, competition and other indirect effects on livestock health, damage to infrastructure (e.g. fencing) and nuisance to farmers. The impacts that wild deer exert on agricultural plants include grazing and browsing damage to horticulture (Fig. 1b), vineyards and forestry plantings. There is little quantitative evidence of these impacts, but abundant anecdotal information (Bowman 2014). A 2008 study into agricultural impacts of deer across Victoria revealed that the main reasons given by landholders for wanting to control deer concerned prevention of damage to the following agricultural values: pasture, fruit, grapevines, vegetables (especially potatoes), pine and other trees, native tree revegetation, and flowers and foliage (Lindeman and Forsyth 2008). Field assessments of Gippsland farms reporting deer damage confirmed browsing damage to tree plantings and broken branches on fruit trees (Fig. 1b).

There is potential for transmission of infectious diseases and parasites from deer to livestock (Fig. 1c) and this is currently an active field of research in south-eastern Australia (Cripps *et al.* 2019). Deer can transmit important livestock viruses such as pestivirus (Huaman *et al.* 2020) and significant livestock parasites such as liver fluke *Fasciola hepatica* (Jenkins *et al.* 2020). In addition, though not documented in published studies conducted in Victoria, farmers may be inconvenienced by hunters illegally accessing private property to hunt deer (Jagnow *et al.* 2006). In Victoria, permits are no longer required for landholders to control deer where they are causing damage, but, in 2008, farmers seeking permits to cull deer were concentrated in Gippsland (Lindeman and Forsyth 2008), no doubt reflecting deer distribution at that time.

Public health impacts

Wild deer can exert negative impacts on human health values via four main pathways. First, vehicle collisions involving deer are becoming increasingly common across Victoria (Ang *et al.* 2019; Davies *et al.* 2020) (Fig. 1d) and

may occur wherever deer populations exist. Second, in the east of Melbourne, pollution of drinking water sources with parasites such as *Cryptosporidium* and *Giardia* occurs via faecal deposition (Cinque *et al.* 2008; Nolan *et al.* 2013; Bennett *et al.* 2015). Third, wild deer are involved in the transmission of zoonotic diseases, such as Q-fever (caused by the bacterium *Coxiella burnetii*) (Davis *et al.* 2016). Overseas, deer are also involved in the transmission of tick-borne zoonotic diseases such as Lyme disease (caused by the bacterium *Borrelia burgdorferi*) in the USA (Kilpatrick *et al.* 2014). This is not currently known to occur in Australia. Finally, deer can cause inconvenience and stress (e.g. through damage to gardens) to residents when populations establish in urban areas (Burgin *et al.* 2015), as is increasingly common on Melbourne's eastern fringe.

Management of impacts

There is a need to manage deer to reduce impacts on biodiversity, agriculture and public health across the state. However, this is costly and complex due to the range of interacting factors that influence deer management. Effective management will require innovative strategies that are cross-tenure and involve a range of stakeholders. Strategies need to be adaptable to meet different objectives, depending on land tenure, ranging from biodiversity conservation and impact mitigation in urban or agricultural settings, to recreation and resource utilisation. Critically, successful management and pragmatic use of limited resources requires an understanding of the ecology of each species and the way in which they interact with, and impact on, natural and production ecosystems. Research that aims to test the effectiveness of management strategies to reduce deer impacts is underway in diverse locations across Victoria, including alpine regions (Ryan-Colton *et al.* 2015; Brown *et al.* 2016), East Gippsland (Edwards and Mills 2014; Davis 2018) and peri-urban Melbourne (Bennett and Davis 2017).

Conclusion

The four species of introduced deer that have established wild populations in Victoria cause a variety of impacts to biodiversity, agriculture

and public health, in a range of locations on public and private land across the state, from parks, reserves and water catchments, to agricultural and peri-urban landscapes. Deer, in particular Sambar and Fallow Deer, are therefore an urgent management issue that requires a strategic response. Although research into deer ecology, impacts and management in Victoria is progressing, further work is required. Priorities include improving understanding of the long-term community- and ecosystem-level consequences of deer impacts in natural systems, the impacts of deer on agriculture and public health, and interactions of these with disturbances such as fire and climate change, as well as development of cost-effective monitoring and management techniques. The resulting knowledge and tools will enable better decisions regarding prioritisation and management strategies for asset protection.

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Insects and Marine Creatures – ABRS: <<http://www.environment.gov.au/biodiversity/abrs/online-resources/fauna/index.html>>.

Birds – Menkhorst P, Rogers D, Clarke R, Davies J, Marsack P and Franklin K (2021) *The Australian Bird Guide* Rev'd Edition (CSIRO: Collingwood).

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